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A Stress Function for Evaluating Strategies for Water Quality Management

Illinois Natural History Survey, Urbana

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A STRESS FUNCTION FOR EVALUATING STRATEGIES FOR WATER QUALITY MANAGEMENT

by

Warren U. Brigham
Illinois Natural History Survey
Urbana, Illinois 61801

Donald L. Hey
Donald L. Hey and Associates
Chicago, Illinois 60657

Grant No. R-805614-01-0

Project Officer

James A. Heidman
Wastewater Research Division
Municipal Environmental Research Laboratory
Cincinnati, Ohio 45268

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16. ABSTRACT

This investigation explored the relationship between biological communities and the physical and chemical conditions in the aquatic environment. Seasonal patterns and the duration and probability of occurrence of chemical conditions and physical events were established via computer modelling. An instantaneous measure of stress was calculated by summing the decimal fractions of the 96-hr lethal concentrations (for bluegills) of each of 21 separate toxicants. These data were calculated over 1-hr time intervals and summarized as a quasi-continuous stress function.

Stress functions were computed for several locations in a test watershed in northeastern Illinois. The functions were related to the diversity and type of fish populations known to exist at each of the locations. The effect of a hypothetical management plan for more stringent wastewater treatment plant standards and control of combined sewer overflows was investigated through use of the model and the resulting stress index was used to estimate the diversity of the fishery which would result if the management plan were implemented.

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the researcher and the user community.

The stress function developed in this investigation provides a methodology for relating the diversity of the fish community to the time varying concentration of twenty-one separate toxicants.

Francis T. Mayo, Director
Municipal Environmental Research
Laboratory

ABSTRACT

This investigation explores the relationship between biological communities and the physical and chemical conditions in the aquatic environment. Seasonal patterns and the duration and probability of occurrence of chemical conditions and physical events are established via computer modelling. An instantaneous measure of stress is calculated by summing the decimal fractions of the 96-hr lethal concentrations (for bluegills) of each of 21 separate toxicants. These data are summarized as a quasi-continuous stress function calculated over 1-hr time intervals. The stress function is related to five distinct biological communities ranging from the most tolerant of pollution to the least tolerant.

Data from a test watershed in northeastern Illinois yielded stress functions from 0.120 to 783.7 (mean 23.02) from a site having no fishes, stress functions from 0.155 to 98.47 (mean 1.038) from a site having a carp population, and stress functions from 0.005 to 0.279 (mean 0.116)) from a site having a bass population.

A hypothetical management plan to reduce the ammonia component at the no fish site was incorporated into the stress function. This plan limited effluent ammonia concentrations to 1.5 mg liter^{-1} during the summer months and 4.0 mg liter^{-1} during winter, eliminated combined sewer overflows, reduced sediment oxygen demand levels substantially, and increased dissolved oxygen levels moderately in treatment plant effluents. Mean stress was reduced by more than an order of magnitude, down to 2.12. This level was still significantly higher than that of the carp site, primarily because of residual chlorine in the effluents from wastewater treatment plants. These results suggest that if a more diverse fishery than carp is desired, further control strategies might need to be implemented.

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SECTION 1

INTRODUCTION

Making and keeping our nation's surface waters fishable and swimmable are admirable goals. In and of themselves, these goals convey an unfettered and reassuring concern for our environment. Although they appear to be clear and simple goals, they are in reality vague and complex. Since they are the key decision criteria of the Federal Water Pollution Control Act Amendments of 1972 (Public Law 92-500), they deserve careful definition.

Yet, these goals are not without economic or social implications. Their achievement undoubtedly will effect billions of dollars of construction; the value of countless acres of land; and the income, health, and well-being of millions of American families. Structural controls, where required, will irreversibly commit capital and energy. While most riparian land values will increase, other land values may decrease due to restricted use. Farm incomes may be reduced and urban taxes increased if sediment and nutrients reaching our streams are to be controlled.

The magnitude of the costs and benefits depend upon the definition of fishable and swimmable. These definitions, however, are not provided by the law. While both are important, the definition of fishable is more restrictive. Providing there is sufficient water and that sources of pathogens are disinfected, swimmability can be assumed if a body of water is fishable. Consequently, the definition of fishable is a subject of major concern.

Is a fishable stream one in which carp are present? Or, is it one from which bass or trout can be caught? Deciding which species are to be encouraged and maintained is the first step toward defining fishable. The decision should be made with an understanding of environmental limitations. Gaining this understanding might best be achieved by determining what fish currently inhabit local streams and lakes and by determining, if possible, what fish inhabited those bodies of water prior to agricultural, industrial, or urban development.

Having selected the desired species, the next part of the definition is derived from their environmental requirements. In order for fish to survive, their food supply must be adequate. This requires the survival of numerous interrelated organisms. The survival of fish and their supporting food chain are dependent upon water chemistry and certain physical conditions in the aquatic environment. Such factors as dissolved oxygen, temperature, flow, velocity, and depth are important. The definition of fishable requires the

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determination of these conditions in space, time, and frequency. Fixed standards such as 5.0 mg/liter dissolved oxygen are not sufficient to insure or determine the survival of the desired fish or food chain. Nor are mean conditions sufficient to indicate survival. Aquatic organisms are effected by the magnitude, duration, timing, and repetition of chemical and physical events within their environment. Further, their requirements are different for different periods of their life cycle. For example, spawning requires one temperature condition, incubation another. A fish often can survive extreme temperatures for periods of short duration, but it may fair less well under moderate increases in temperature which last for long periods of time. Finally, the factors of magnitude, duration, and timing must be placed in perspective. That is to say, the probability of their occurrence must be determined. This latter consideration establishes the level of expectation by which benefits can be compared to costs.

A major assumption of benefit-cost analysis is the assumption that the benefits actually will accrue from implementation of a project. In the case of fishability, the assumption becomes one of assuming that a certain fishery will become possible after implementation of a specific set of water quality management practices. This report introduces the concept of the stress function and demonstrates how it can be used to test the validity of these assumptions concerning the attainment of fishabilty.

The stress function extends our understanding of the interrelationships between existing water quality conditions and the aquatic biota. It has proven useful in comparing the relative effectiveness of proposed solutions to water quality problems and could be adapted to anticipate future water quality problems. In particular, it assigns to the predicted aquatic community a role in evaluating strategies for water quality management.

SECTION 2

CONCLUSIONS AND RECOMMENDATIONS

1. Once calibrated, computer models can be used to simulate the seasonal patterns and the duration and probability of occurrence of chemical conditions and physical events in a river basin.

2. These simulated data may be summarized as a quasi-continuous stress function.

3. Maximum and mean values of the stress function at a given site correlate well with the fish population found there. In general, no fish sites yielded values an order of magnitude higher than sites having carp populations. Carp sites yielded values an order of magnitude higher than bass sites.

4. The stress function demonstrated that a hypothetical water quality management plan for ammonia reduction would produce only a marginal improvement in the fish population at a degraded site. Demonstrations such as this could provide useful information to planners who need to predict benefits which might accrue from alternative strategies for water quality management.

5. This investigation tested the stress function in northeastern Illinois in a river basin having three types of aquatic communities (no fish, carp, and bass sites). The stress function should be tested in higher quality water (pike and trout sites) and should be tested in other regions of the country to determine if the method is generally applicable.

SECTION 3

OBJECTIVES

The purpose of this investigation was to explore and establish the relationship between biological communities and the chemical and physical conditions in the aquatic environment. Seasonal patterns of chemical concentrations and physical events were established, as were their duration and probability of occurrence. This information was summarized as a quasi-continuous stress function calculated over relatively short time intervals (1 hr). The stress function was related to five distinct biological communities ranging from the most tolerant of pollution to the least tolerant. Each community was identified as to its characteristic, dominant, or predominant fish species. Data from a test watershed was used to demonstrate the relationship between the stress function and these biological communities. Finally, a stress function was calculated for the same test watershed with the assumption that a set of water quality management goals had been met. This stress function was used to evaluate the effectiveness of these goals in terms of fishability.

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SECTION 4

THE STRESS FUNCTION

CONCEPT

Water quality has become an increasingly sensitive and multifaceted issue over the last 30 years. The physical and chemical properties of water depend upon the characteristics of surface and groundwater runoff, flow, hydrology, and sediment-water interactions. These characteristics then establish areas in which certain species of aquatic organisms exist. Section 101(a) of the Federal Water Pollution Control Act Amendments of 1972 establishes a national goal of establishing and/or maintaining healthy populations of aquatic organisms without imposing unnecessarily stringent limitations upon dischargers or users of land adjacent to streams and lakes. It is important to establish which pollutants and at what levels these pollutants interfere with aquatic life. It also is important to then establish the degree of control necessary to restore or maintain aquatic life at a desired level.

Combined actions of the public and state and federal agencies have produced many advances in water resource management practices. Concurrently, numerous indices have been developed for estimating water quality. Among these are

- Single constituent indicators
- Criteria-based and standards-based indicators
    - Prevalence of pollution index
    - USEPA Region VIII index
    - USEPA Region X index
    - MITRE prevalence, duration, and intensity index
- Judgmental multi-parameter indicators
- Empirical multi-parameter indicators
- Lake indicators
    - National Eutrophication Survey single-parameter index
    - National Eutrophication Survey trophic index
    - Dobson index
- Aquatic life indicators
    - "Indicator" species
    - Bioassay
    - Shannon-Weaver (species diversity) index
    - Equitability
- Water-use indicators
- Perception-based indicators
- Point-source indicators
- Non-point source indicators

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In this context, water quality indices represent a grading system for the comparison of various waters and much effort is directed at quantifying "good" versus "bad" and the values between these extremes. The index becomes a summation of the individual effects of the components used to develop the index. This attribute of an index allows direct comparison of the overall quality of different waters even though the concentration ranges of the individual components may be very different. The water quality index also is a useful tool in bridging the information gap between technical and non-technical personnel.

Most existing water quality data networks are too sparse and the sampling intervals too infrequent to analyze areal distribution of water quality and diurnal, weekly, and/or monthly variations. Frequency-duration analysis also is usually impossible. Therefore, a water quality index should be based upon a computer model capable of extending existing data. Ideally, this model should relate the body of water to its watershed and should simulate the interaction between meteorologic, hydrologic, water quality, and biological processes. Such a model has been developed by Hydrocomp, Inc., and has been used by the Northeastern Illinois Planning Commission to simulate water quality in streams within its planning boundries. The components of this model will be discussed in detail below.

This report introduces the concept of the stress function as a means of expressing the interrelationships between water quality conditions and the aquatic biota. It is based, in part, on the bluegill toxicity index developed by Drs. Richard E. Sparks and Kenneth S. Lubinski of the Illinois Natural History Survey. The toxicity index represents an instantaneous summation of the concentrations of up to 20 toxicants relative to their acute lethal effects upon the bluegill, *Lepomis macrochirus*. The stress function, however, is not an instantaneous value. It is a quasi-continuous time-series obtained by marrying toxicity index calculations to the output of the Hydrocomp model. The stress function is constructed from the given information regarding the physical and chemical conditions of the stream (*i. e.*, point sources, non-point sources, and instream physical and chemical interactions). Thus, each function represents the signature of the stream reach and reflects the stress that is endured by the resident aquatic community.

In order to quantify the association between stress and the aquatic community, a number of statistical characteristics can be developed to compare stress functions from stream reaches supporting different biological communities. Four such characteristics can be taken from the stress function itself: maximum, minimum, mean, and standard deviation of stress. As will be seen later, maximum and mean values increase from intolerant to tolerant communities. In fact, the mean stress function (derived from hourly values calculated for a 3-year period) appears to change by more than an order of magnitude as one moves from a bass to a carp to a no fish community.

Other characteristics of the stress function are derived from the relationship between the stress function and an arbitrary reference point here designated as the stress index. The stress index is not intended to reflect acute, critical, or chronic thresholds of stress. It is merely a reference level about which to measure the statistical properties of the stress function. Note that a greater portion of the stress function would be expected to lie above the stress index for the successive communities from bass to carp to no fish. In the same order of succession, the mean stress should increase and perhaps finally exceed the stress index. Also important is the fact that the duration between excursions above the stress index should tend to shorten with an increase in mean stress. The stress function associated with bass would have much longer periods of time between excursions above the index as compared to the stress function associated with no fish.

Additional characteristics of the stress function may be described relative to the stress index. One such characteristic is the frequency of excursions above the index. While excursions above the stress index may be important to some organisms, particularly where these excursions are extreme, the duration of these events may be just as important. Consequently, another statistic which will need to be derived is the duration of positive events. Finally, the time spent between positive excursions might be used to represent a recovery period. If positive excursions occur too frequently and at short intervals, this may lead to the decline of the aquatic community. Consequently, the duration of negative events (_i. e._, the time spent between positive events) will be measured.

The sections which follow describe in detail the four principal components of the computer model which was used to generate 3 years' of hourly values for the stress function at three sites in a test watershed in northeastern Illinois.

COMPONENTS AND PROCEDURES

The Nonpoint Source Model, NPS

The NPS Model (available from Hydrocomp, Inc., Palo Alto, CA) was developed as a tool for the evaluation and analysis of nonpoint pollution problems. The model continuously simulates hydrologic processes (including snow accumulation and melt); erosion processes; and pollutant accumulation, generation, and transport from the land surface. Sediment and sediment-like material are used as the basic indicator of nonpoint pollutants. These erosion processes are simulated separately on both pervious and impervious areas. Pollutant loadings are determined by multiplying the resulting sediment discharge by "potency factors" representing the pollutant strength of the sediment. The model simulates the processes that determine nonpoint pollution and is applicable to urban, agricultural, forested, and construction areas.

The NPS Model is a pollutant loading model in that it simulates the total input or pollutant loading to a stream channel or waterbody. Although the hydrologic algorithms simulate all runoff components (surface runoff, interflow, groundwater flow), the present version of the model dynamically simulates only pollutant processes on the land surface. Subsurface and groundwater pollutant loads are determined by assuming constant (or monthly variable) pollutant concentrations for interflow and groundwater. In-stream processes are not represented, and all pollutants are assumed to be conservative within the 15-min simulation interval of the model.

The model is composed of three major subroutines: MAIN, LANDS, and QUAL. The operational flowchart of the NPS Model (Fig. 1) demonstrates the sequence of computation and the relationships between the various subroutines. MAIN, the master or executive subroutine, reads model parameters and meteorologic data, initializes variables, monitors the passage of time, calls the LANDS and QUAL subroutines, and prints monthly and yearly output summaries. LANDS, based upon the Stanford Watershed Model, simulates the hydrologic response of the watershed and the processes of snow accumulation and melt. LANDS is described in detail in a later section of this report. The QUAL subroutine simulates erosion processes, sediment accumulation, and pollutant transport from the land surface. The erosion algorithms are derived from research work on simulating agricultural runoff.

The processes on pervious areas simulated in the QUAL subroutine include (1) net daily accumulation of sediment by dustfall and human activities, (2) detachment of sediment aggregates by raindrop impact into fine sediment material, and (3) transport of sediment fines by overland flow. On pervious areas, detachment heavily outweighs dustfall and accumulation from land surface activities. Hence, the accumulation algorithm will be discussed in the section on impervious areas where it is the sole source of surface sediments. However, accumulation also is simulated on pervious areas.

Soil fines detachment:

$$RER(t) = (1 - COVER(T)) * KRER * PR(t)^{JRER}$$

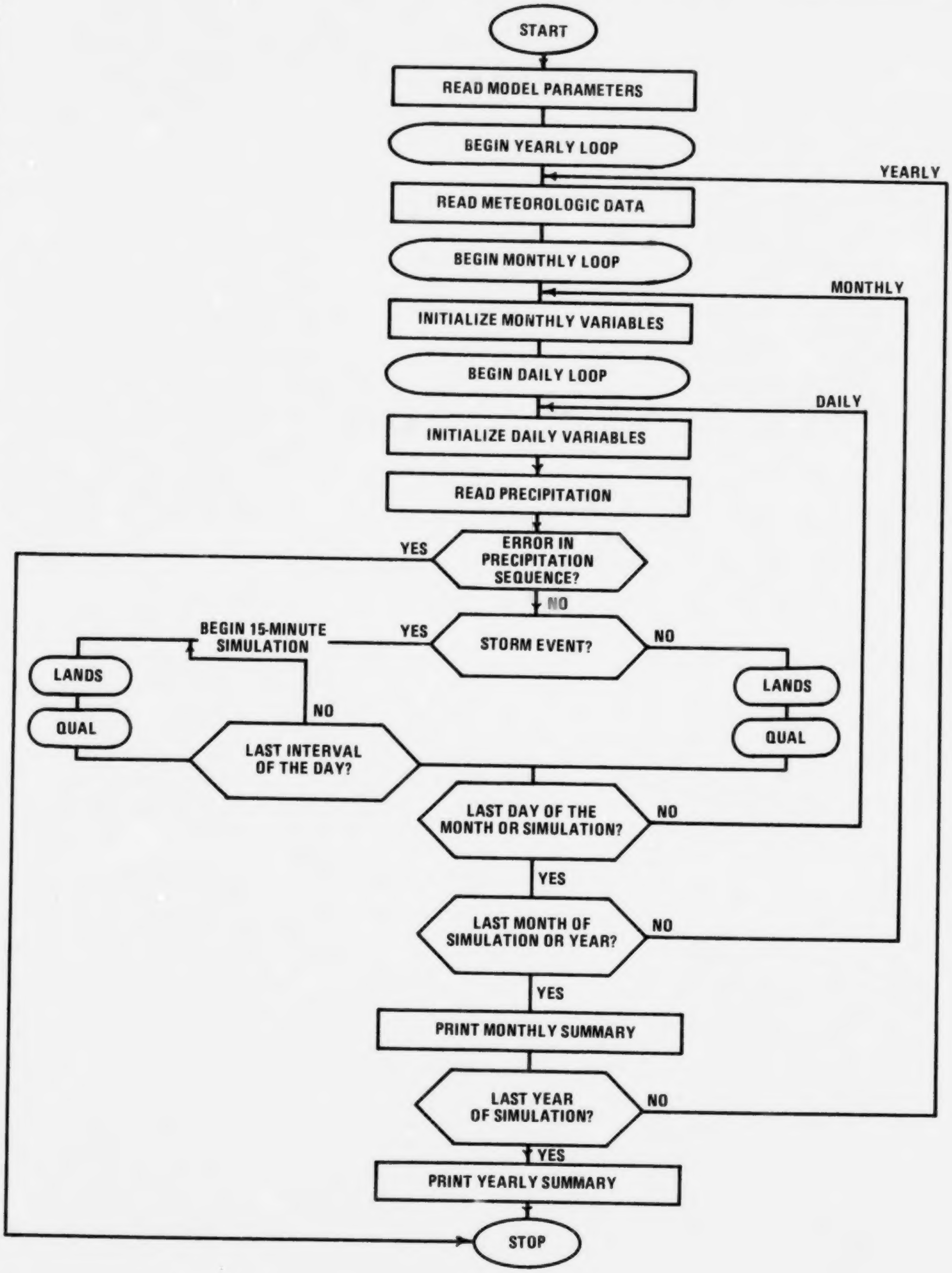
START
READ MODEL PARAMETERS
BEGIN YEARLY LOOP
YEARLY
READ METEOROLOGIC DATA
BEGIN MONTHLY LOOP
MONTHLY
INITIALIZE MONTHLY VARIABLES
BEGIN DAILY LOOP
DAILY
INITIALIZE DAILY VARIABLES
READ PRECIPITATION
YES
ERROR IN PRECIPITATION SEQUENCE?
NO
BEGIN 15-MINUTE SIMULATION
YES
STORM EVENT?
NO
LANDS
QUAL
NO
LAST INTERVAL OF THE DAY?
LANDS
QUAL
LAST DAY OF THE MONTH OR SIMULATION?
NO
YES
LAST MONTH OF SIMULATION OR YEAR?
NO
YES
PRINT MONTHLY SUMMARY
LAST YEAR OF SIMULATION?
NO
YES
PRINT YEARLY SUMMARY
STOP

Figure 1. Flow diagram for the NPS Model.

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$$SRER(t) = SRER(t - 1) + RER(t)$$

Soil fines transport:

$$SER(t) = \begin{cases} KSER*OVQ(t)^{JSER} & \text{for } SER(t) < SRER(t) \\ SRER(t) & \text{for } SER(t) \geq SRER(t) \end{cases}$$

$$ERSN(t) = SER(t)*F$$

where
RER(t) = soil fines detached during time interval t, tons/acre
COVER(T) = fraction of surface land cover as a function of time, T during the year
KRER = detachment coefficient for soil properties
PR(t) = precipitation during the time interval, in
JRER = exponent for soil detachment
SER(t) = transport of fines of overland flow, tons/acre
KSER = coefficient of transport
JSER = exponent for fines transport by overland flow
SRER(t) = reservoir soil fines at the beginning of time interval, t, tons/acre
OVQ(t) = total overland flow occurring during the time interval, t, in
F = fraction of overland flow reaching the stream during the time interval, t
ERSN(t) = sediment loss to the stream during the time interval, t, tons/acre

In the operation of the algorithms, the soil fines detachment during each 15-min interval (RER) is calculated and added to the total fines storage (SRER). Next, the total transport capacity of the overland flow (SER) is determined. Sediment is assumed to be transported at capacity if sufficient fines are available; otherwise, the amount of fines in transport is limited by the fines storage (SRER).

The sediment entering the waterway in the time interval is calculated from the fraction of total overland flow that reaches the stream. An overland flow-routing technique determines the overland flow contribution to the stream in each time interval. After the fines storage (SRER) is reduced by the actual sediment entering the stream (ERSN), the algorithms are ready for simulation of the next time interval. Thus, the sediment that does not reach the stream is returned to the fines storage and is available for transport in the next interval. The land cover variable, COVER(T), represents the fraction of the land surface effectively protected from the kinetic energy and detachment capability of rainfall. Monthly cover values as of the first day of the month are specified by the user. The NPS Model interpolates linearly between the monthly values to evaluate land cover on each day. Figure 2 demonstrates the land cover function in the NPS Model.

In essence, the land cover function is the key to differentiating erosion

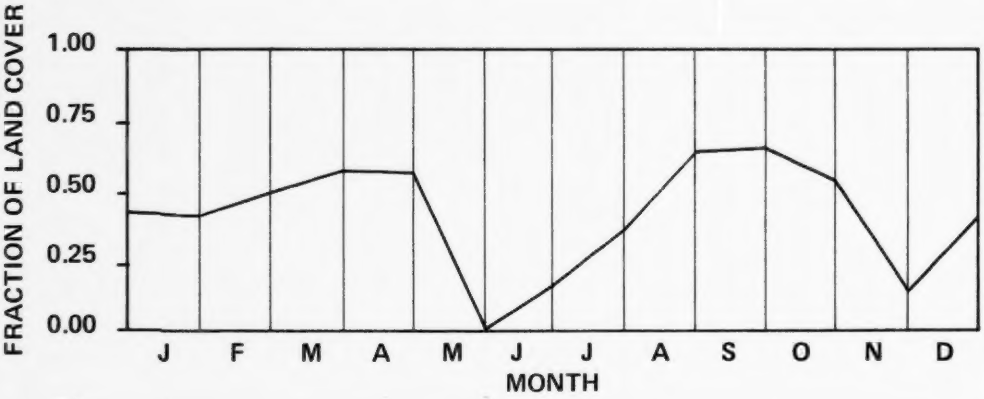

Figure 2. An example of the land cover function in the NPS Model.

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rates on different land uses. Agricultural, silvacultural, and construction areas will have highly variable land cover, with portions of the land surface completely exposed during certain seasons of the year. The land cover function in Figure 2 is typical for an agricultural watershed. Storm events occurring where the land is exposed can produce severe sediment loss. On the other hand, sediment loss can be low from the pervious portion of urban areas (*e. g.*, lawns, parks, golf courses, etc.) that have a reasonably constant and complete vegetative cover. The kinetic energy of rainfall is effectively dissipated by the land cover with values of 90% to 95% of the area. Thus, judicious use of the land cover function in the NPS Model will allow simulation of various land surface conditions.

The timing and severity of tillage operations have a controlling effect upon the sediment loss from an agricultural watershed. The effect of tillage on sediment processes is to increase the mass of fine soils available for transport. The NPS Model allows the user to specify the dates of tillage or land-surface disturbing operations. For each of these dates (TIMTIL), the user must specify a new detached soil fines storage (SPERTL) resulting from the operation. At the beginning of each day when tillage occurs, the model resets the fines storage to the new value. The amount of fines storage produced by different tillage operations is related to soil characteristics and the depth and extent of the operation. This option and the land cover function can be used jointly to represent alternate agricultural practices (for example, winter cover crops, fall plowing, conservation tillage) and areas undergoing construction.

The important processes on impervious areas are the accumulation of pollutants on the land surface and the transport of pollutants by overland flow. Accumulation of dust, dirt, debris, and other contaminants from streets, roads, and parking lots is the major source of nonpoint pollutants on impervious areas. The composition of these pollutants is similar to sediment, and is often measured as total solids (suspended and settleable). Thus, these pollutants are simulated as sediment on impervious areas. Rates of sediment accumulation on impervious areas are a function of land use, street-cleaning practices, and climatic factors such as wind and rainfall.

To evaluate the amount of sediment on the watershed prior to each event, the effects of the non-runoff removal processes must be determined and incorporated into the accumulation function. The accumulation function simulates the net accumulation of sediment, that is, the difference between accumulation and removal by mechanisms other than runoff. The major removal processes of concern are street cleaning, and entrainment and transport by wind. The accumulation function in the QUAL subroutine is

$$TS(T) = TS(T-1)*(1-R) + ACCI$$

where $TS(T)$ = sediment on the impervious land surface at time T

$TS(T-1)$ = sediment on the impervious land surface at time $T-1$

R = fraction of sediment removed daily

$ACCI$ = daily accumulation rate of sediment

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R and ACCI are dependent upon land use and season of the year. The formulation for pervious areas is identical to the above, with separate accumulation and removal rates and separate sediment storage.

In the operation of the QUAL subroutine, the accumulation function is performed each day that a storm does not occur. Thus, as time between storm events increases, the accumulated sediment approaches the limiting value. The limit of daily change is represented by

$$\Delta TS = -TS(T)*R + ACCI$$

and at equilibrium $\Delta TS = 0$

The limiting value of accumulated sediment is represented as

$$TS(T) = ACCI/R$$

This shows that the limiting value of TS(T) is the daily accumulation rate divided by the daily removal rate. The maximum accumulation would be 1/R in terms of days of accumulation.

Sediment transport from impervious areas is analagous to the same process on pervious areas. It is represented as follows:

$$TSS(t) = \begin{cases} KEIM*OVQ(t)^{JEIM} & \text{for } TSS(t) < TS(t) \\ TS(t) & \text{for } TSS(t) > TS(t) \end{cases}$$

$$EIM(t) = TSS(t)*F$$

where
TSS(t) = sediment transport during time interval, t, tons/acre
OVQ(t) = impervious area overland flow occurring in time interval, t, in
KEIM = impervious area coefficient of transport
JEIM = impervious area exponent of transport
TS(t) = reservoir of deposited sediment on impervious areas, tons/acre
F = fraction of impervious overland flow reaching the stream in time interval, t
EIM(t) = sediment loss to the stream from impervious area in time interval, t, tons/acre

As with pervious areas, sediment transport is limited in each time interval by the availability of deposited sediment. Total sediment output to the stream per acre impervious area is proportional to the fraction of total overland flow entering the stream during the time interval.

The operation of the QUAL subroutine, as illustrated in Figure 3, consists of two alternate loops, each one iterated with a different frequency, depending upon the rainfall and runoff conditions transferred from the LANDS subroutine. At the beginning of each simulation day, the MAIN subroutine determines whether or not a storm has occurred on that day; daily rainfall

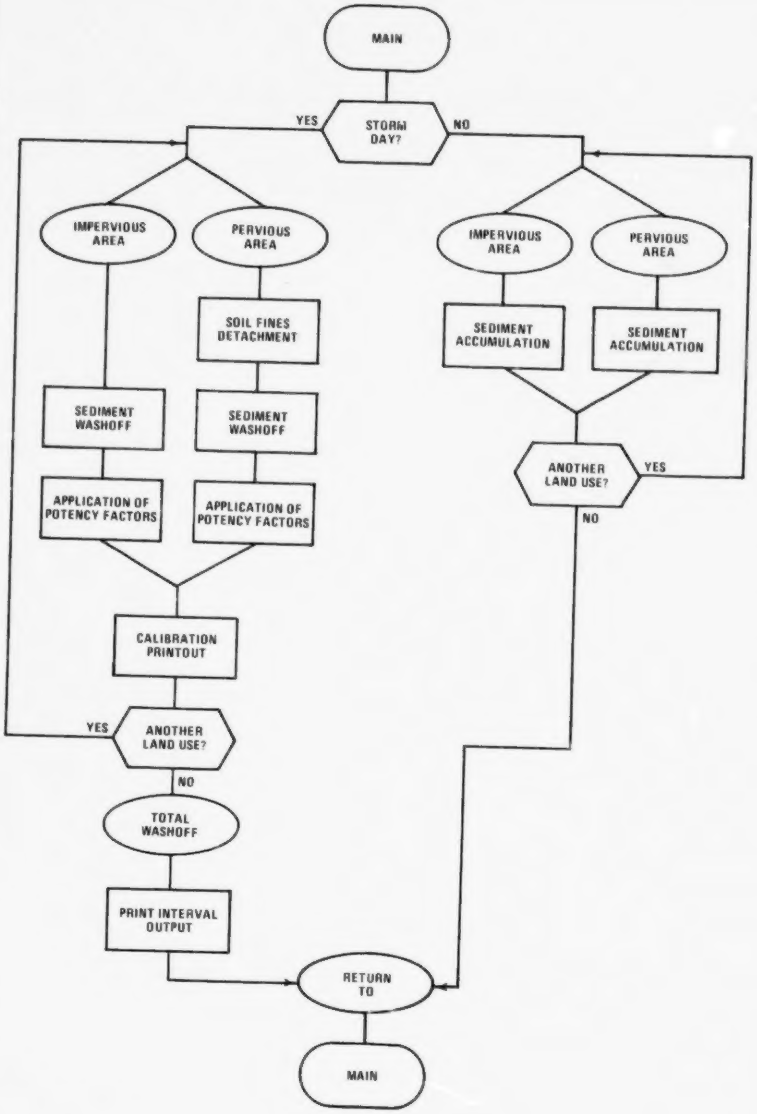
MAIN
STORM DAY?
YES
NO
IMPERVIOUS AREA
PERVIOUS AREA
SOIL FINES DETACHMENT
SEDIMENT WASHOFF
SEDIMENT WASHOFF
APPLICATION OF POTENCY FACTORS
APPLICATION OF POTENCY FACTORS
CALIBRATION PRINTOUT
YES
ANOTHER LAND USE?
NO
TOTAL WASHOFF
PRINT INTERVAL OUTPUT
IMPERVIOUS AREA
PERVIOUS AREA
SEDIMENT ACCUMULATION
SEDIMENT ACCUMULATION
ANOTHER LAND USE?
YES
NO
RETURN TO
MAIN

Figure 3. Flow diagram for the QUAL subroutine.

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and/or the occurrence of overland flow indicate a storm day. Whenever a storm day occurs, both the LANDS and QUAL subroutines are iterated sequentially throughout the whole day at 15-min intervals (96 times). Otherwise, the nonstorm path is activated resulting in only one call to the LANDS and QUAL subprograms. In this case, the role of the QUAL algorithm is limited to the evaluation of the daily increments of sediment available for transport from the pervious and impervious lands. The calculations are carried out iteratively for each of the land uses defined by the input data. The factors considered are the daily accumulation rate in mass per unit area (lb/acre) and the removal effect representing the percent of sediment loss due to wind, street cleaning, and other factors not related to storm runoff. Both accumulation and removal rates must be specified separately for the pervious and impervious areas.

The storm-day loop of the QUAL subroutine includes the analytical representations of sediment fines generation, sediment washoff, and pollutant washoff from pervious and impervious areas. Simulation of these processes is carried out for each land use within the watershed. The aggregate quantities of the washed-off sediments and pollutants are summed to yield the total mass and the equivalent concentration of pollutants in the overland flow.

The Hydrologic Model, LANDS

Relative to the aquatic environment, streamflow is the hydrologic response of greatest interest. Figure 4 represents the major elements of the hydrologic cycle which affects streamflow. These elements are rainfall; snow and snowmelt; infiltration; soil moisture storage; evapotranspiration; surface runoff; interflow; groundwater; and municipal, industrial, and agricultural discharges. Each element is represented in the model by one or more mathematical relationships. Each relationship has one or more parameters determined by watershed characteristics. For example, infiltration capacity is represented as a function of soil moisture storage:

$$\bar{f} = \mathrm{INFILTRATION}/(\mathrm{LZS}/\mathrm{LZSN})$$

where

$\bar{f}$ = time-variable infiltration capacity

INFILTRATION = a parameter relating to soil type

LZS = soil moisture storage at anytime as computed by the model

LZSN = a parameter defining the normal soil moisture storage capacity

$\bar{f}$ is a dependent variable, LZS is an independent variable, and INFILTRA-

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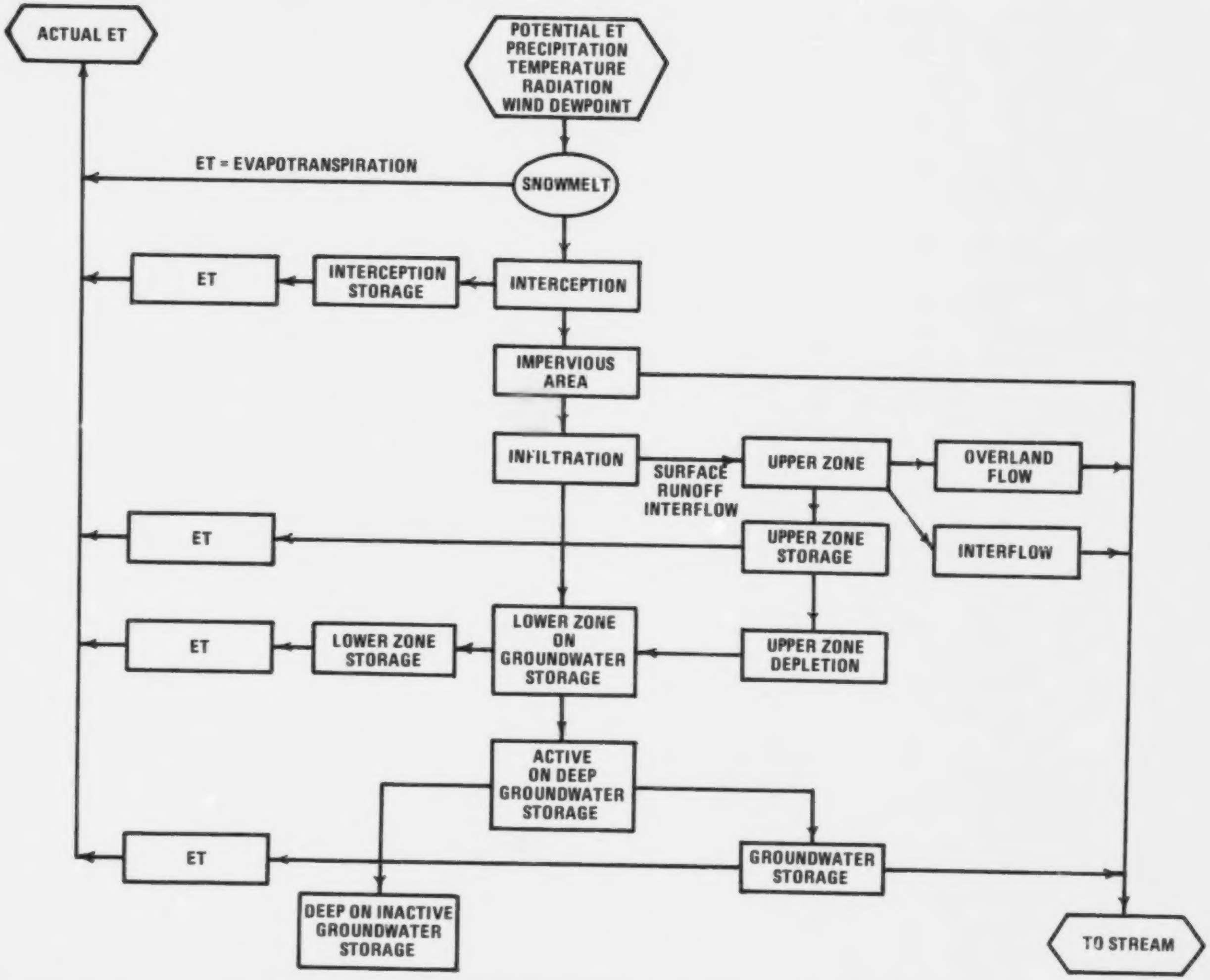
POTENTIAL ET
PRECIPITATION
TEMPERATURE
RADIATION
WIND DEWPOINT
ACTUAL ET
ET = EVAPOTRANSPIRATION
SNOWMELT
INTERCEPTION
INTERCEPTION STORAGE
ET
IMPERVIOUS AREA
INFILTRATION
SURFACE RUNOFF INTERFLOW
UPPER ZONE
OVERLAND FLOW
INTERFLOW
UPPER ZONE STORAGE
ET
UPPER ZONE DEPLETION
LOWER ZONE ON GROUNDWATER STORAGE
LOWER ZONE STORAGE
ET
ACTIVE ON DEEP GROUNDWATER STORAGE
GROUNDWATER STORAGE
ET
DEEP ON INACTIVE GROUNDWATER STORAGE
TO STREAM

Figure 4. Flow diagram for the LANDS Model.

TION and LZSN are parameters.

An annotated list of parameters is given in Tables 1 and 2. Most of these values can be determined directly from the physical characteristics of the watershed. The parameter values relating to land slope and land cover, for example, can be determined directly from topographic and land-use maps. However, other parameter values, such as infiltration, are derived in calibration. Assuming an initial value for infiltration based upon the soil types in the watershed, 5 years of hourly streamflow are simulated and compared to observed values. If there is a substantial difference between the simulated and recorded flows which could be corrected with a different value of infiltration, the change is made and the simulation is repeated.

The data that are required for calibration can be divided into two sets: steady-state and time-series. The steady-state data include land cover, soil type, surface slope, stream gradient, and stream cross section. These data are used to establish the values of the parameters of Tables 1 and 2. If the physical condition of the watershed remains the same, the parameters are constant over the calibration period. However, if substantial changes in land cover occur during the calibration period, the land cover representation is changed in the model to reflect the physical changes.

Time-series data include hourly precipitation, daily maximum and minimum air temperatures, wind movement, dew-point temperature, cloud cover, radiation, streamflow, and semi-monthly evaporation. The meteorologic data are input to the model. The streamflow data are used to check the output of the model.

The model transforms the input data into streamflow using a moisture accounting procedure. A generalization of this is:

$$Q = P - L + \Delta S$$

where Q = streamflow

P = precipitation plus snow melt

L = losses due to evaporation, transpiration

ΔS = change in soil moisture

The roles of streamflow, precipitation, and evaporation data are apparent from this equation. The other data are used to compute the rates of snow accumulation and melt. When snowmelt occurs it is added to precipitation in the water balance.

A successful calibration requires that the simulated streamflows closely resemble the corresponding measured values. The correspondence must exist in both space and time. Further, there must be a correspondence between the statistical characteristics of the observed and simulated streamflows. Recorded and simulated streamflows should correspond at the outlet of the watershed as well as at interior locations along tributaries and at stream

TABLE 1. DEFINITIONS OF TERMS USED IN THE LANDS MODEL

LAND	
K1	Ratio of average segment rainfall to average gage rainfall
A	Impervious area (fraction)
EPXM	Interception storage (maximum value)
UZSN	Nominal upper zone soil moisture storage
LZSN	Nominal lower zone soil moisture storage
K3	Actual evaporation rate parameter
K24L	Seepage to 'deep' groundwater
K24EL	Evaporation from perched groundwater
INFILTRATION	Infiltration
INTERFLOW	Interflow
L	Length of overland flow
SS	Overland flow slope (ft/ft)
NN	Manning's N for overland flow
IRC	Daily interflow recession rate
KV	Groundwater recession, variable rate
KK24	Groundwater recession, constant rate
SNOW	
RADCON	Radiation melt parameter
CONDS-CONV	Convection melt parameter
SCF	Snow correction factor to gage record
ELDIF	Elevation difference (gage to segment)
IDNS	Initial density of new snow
F	Forest cover
DGM	Daily ground melt (inches)
WC	Water content of snowpack maximum
MPACK	Snowpack at complete areal coverage
EVAPSNOW	Snow evaporation parameter
MELEV	Mean watershed segment elevation (ft)
TSNOW	Upper limit of temperature at which precipitation is snow

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TABLE 2. DEFINITIONS OF TERMS USED IN THE CHANNEL SUBROUTINE, LANDS MODEL

REACH	Reach number
LIKE	Reach number that has an identical cross section
TYPE	The type of channel: RECT: Trapezoidal channel cross section; CIRC: Circular conduit; IMAG: Feeder reach without routing; DAM_: Reservoir
TO#	Reach number to which the reach is tributary
SEG#	Land surface segment that contains the reach
LEN	Length of the reach in miles
AREA	Local area tributary to the reach in sq. miles
UPSTR	Upstream channel bottom elevation in the reach
DNSTR	Downstream channel bottom elevation in the reach
W1	Incised channel bottom width in feet for trapezoidal channels, or the diameter in inches for circular channels
W2	Incised channel top width in feet for trapezoidal channels or Manning's n for circular channels
DEPTH	Incised channel depth in feet
S-FP	Transverse slope of the flood plain in feet per foot
N-CH	Manning's n for the incised channel
N-FP	Manning's n for the flood plain

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junctions. Such temporal considerations as annual, monthly, and mean daily water yield need to be compared. The statistical measures to be checked are flow duration and variance.

The accuracy of the hydrologic model is verified partially by the tests of calibration. However, the model should be able to reproduce streamflow for a period other than that used for calibration, with equal fidelity, without modifying parameters, except those related to changes in land cover.

The ability of the hydrologic model to produce results that satisfy these tests involves three considerations: model error, data error, and calibration error. Model errors result when one or more of the mathematical relationships do not describe adequately the prototype process. The results of numerous applications and tests indicate that this error for the model is very small.

Data errors are of two types: measurement and random. Measurement errors occur when the recording instrument malfunctions or the observer misreads the instrument and no value (or an incorrect value) is recorded. In spite of every attempt to correct or avoid these cases, some residual error always remains. Random errors are always present in the measurement of climatic conditions, particularly in the measurement of precipitation. A single rain gage rarely represents the true precipitation over a watershed. Even the use of three or four rain gages does not insure precise representation of the spatial and temporal variations of snow and rainfall. The parameter K1 compensates for this, in part. However, data errors are usually random, and over a period of months, tend to cancel each other. It generally is not possible to perfectly emulate the hour-by-hour observed flows because of random errors. Yet, the probability distribution of hourly, daily, and monthly or annual total flows can be reproduced with small error.

Calibration errors occur when incorrect parameter values have been chosen. They can lead to persistent bias where simulated flows are too high or too low, or they can lead to sporadic bias when the conditions of an infrequent phenomenon are misrepresented. Calibration errors are minimized by simulating and comparing short-interval, long-term streamflow series.

The Instream Water Quality Model, QUALITY

The QUALITY model is linked already to the LANDS model described above. The overland, shallow subsurface, and groundwater flows are simulated by LANDS and used by QUALITY to simulate pollutant washoff and instream physical, chemical, and biological processes. Since both QUALITY and LANDS are mathematical models representing aquatic phenomena, the process of calibrating these modules is similar.

The calibration of QUALITY is divided into two phases: land surface and instream. In each phase values are established for the parameters associated with the mathematical relationships defining that part of the aquatic environment. The parameter definitions for the washoff phase are given in Table 3. Tables 4 and 5 give the parameter definitions for the instream phase. The values of these parameters are based on physical measurements of land surface, land use activities, streams and reservoirs, and calibration. Certain param-

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TABLE 3. DEFINITIONS OF TERMS USED IN THE WASHOFF SUBROUTINE, QUALITY MODEL

PARAMETER	DEFINITION
SEG	Segment Number
CM	Calendar month for which loading rates apply
INITI	Initial surface loading on impervious area in LBS/AC
INITP	Initial surface loading on pervious area in LBS/AC
YI	Loading rate on impervious area in LBS/AC/DAY
LLI	Loading limit on impervious area in days
YP	Loading rate on pervious area in LBS/AC/DAY
LLP	Loading limit on pervious area in days
CONC	Subsurface concentration
SMOOTH	Averaging coefficient for subsurface temperature
OFFSET	Offset parameter for mean subsurface water temperature
RIMP	Washoff coefficient for impervious area in 1/inch
RSUR	Washoff coefficient for pervious area in 1/inch

TABLE 4. DEFINITIONS OF TERMS USED IN THE CHANNEL SUBROUTINE, QUALITY MODEL

NETWORK	
REACH	Reach number
LIKE	Reach number that has an identical cross-section
TYPE	The type of channel: PHBE: Trapezoidal channel cross section RESR: Reservoir IMAG: Feeder reach without routing
N	Number of layers in reservoir
TRIB_TO	Reach number to which the reach is tributary
SEGMT	Segment number of primary tributary segment
LENGTH	Length of the reach in miles
TRIB_AREA	Local area tributary to the reach in sq miles
EL_UP	Upstream channel bottom elevation in the reach
EL_DOWN	Downstream channel bottom elevation in the reach
W1	Incised channel bottom width in feet for trapezoidal channels or the top layer in reservoirs
W2	Incised channel top width in feet for trapezoidal channels or the top layer in reservoirs
H	Incised channel depth in feet
S-FP	Slope of the flood plain in feet per foot
N-CH	Manning's n for the incised channel
N-FP	Manning's n for the flood plain
LKROUTE	
RCH	Reach number containing reservoir
KC	Storage constant when top layer volume less than bankfull volume
HEXC	Discharge exponent when top layer volume less than bankfull volume
KF	Storage constant when top layer volume greater than bankfull volume
HEXF	Discharge exponent when top layer volume greater than bankfull volume
VB	Bankfull volume of top layer in acre-feet
VL	Volume of top layer in acre-feet below which no discharge occurs

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TABLE 4 (continued)

TRIBAREA

RCH	Reach number
SEGMT1	Segment number of primary tributary segment
A1	Impervious area SEGMT1
A2	Pervious area SEGMT1
SEGMT2	Segment number of second tributary segment
A3	Impervious area SEGMT2
A4	Pervious area SEGMT2
SEGMT3	Segment number of third triburary segment
A5	Impervious area SEGMT3
SEGMT4	Segment number of fourth tributary segment
A7	Impervious area SEGMT4
A8	Pervious area SEGMT4

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TABLE 5. DEFINITIONS OF TERMS USED IN THE INSTREAM SUBROUTINE, QUALITY MODEL

QUALITY	
RCH	Reach number
LIKE	Reach number of reach with identical reaction rates
KBOD	BOD decay coefficient in 1/hr at 20°C
KSET	BOD settling rate in ft/hour
KDO	Reaeration correction factor
KEXP	Exposure factor
KSA	Surface area factor
BASEXT	Base extinction coefficient per foot
KNH320	Ammonia oxidation rate in 1/hour at 20°C
ABENT20	Benthal oxygen demand in mg oxygen/square meter/hour at 20°C
BOTTOM	
RELE1B	BOD aerobic release rate in mg BOD/square meter/hour
RELE2B	BOD anaerobic release rate in mg BOD/square meter/hour
RELE1P	Phosphate aerobic release rate in mg P/square meter/hour
RELE2P	Phosphate anaerobic release rate in mg P/square meter/hour
RELE1N	Ammonia aerobic release rate in mg N/square meter/hour
RELE2N	Ammonia anaerobic release rate in mg N/square meter/hour
LANDS	
KEVAP	Evaporation coefficient
KCOND	Conduction coefficient
KATRAD	Atmospheric long wave radiation coefficient
WATERSHED	
ALPHA	Advection averaging coefficient
ALRAT	Ratio of chlorophyll a to phosphorous in algae
OQ	Photosynthetic oxygen coefficient
SINK	Algal sinking rate in reservoirs
SINKC	Algal sinking rate in rivers
TETNIF	Nitrification temperature correction factor
THETBOD	BOD oxidation temperature correction factor
NONREF	Biodegradable fraction of the organic material

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eters, such as the BOD decay rate (KBOD), may be adjusted until the response of the model closely emulates the water quality of the receiving stream.

QUALITY is a synthesis of the current, quantitative knowledge of the aquatic environment. This quantification is both an asset and a limitation. The model structure permits an evaluation of the interactions between climate, land use activities, pollutants, and water quality. The model is limited by the assumptions which enable the representation of these interactions. Used properly, QUALITY is a valuable tool, but for proper use, its assets and limitations need to be understood.

QUALITY simulates accumulation and washoff of pollutants from the land surface and pollutant inflow from groundwater (both nonpoint sources); discharges from municipal, industrial, and agricultural sources (point sources); and pollutant inflow from upstream reaches. The flows and associated pollutants are routed through the receiving drainage system. During routing, the flows and pollutants are subjected to dilution.

Within each reach, whether free flowing or impounded, the pollutant concentration is assumed to be distributed uniformly. Reservoir stratification is represented with as many as nine layers, each of which can have a different pollutant concentration. Pollutants can be transferred between layers.

Land surface washoff is represented in QUALITY by three hourly time series: IMPRO, OLFRO, and SUBRO. Each time series contains flow and quality components. The flow component is an output from LANDS. The quality components are temperature, dissolved oxygen, BOD, ammonia, nitrite-nitrate, and phosphate. They are related to the flows according to land cover, land use, and soil conditions. IMPRO is the washoff time series for impervious land surfaces; OLFRO represents pervious land surfaces, such as cultivated or grassy areas; and SUBRO represents goundwater flow. The flow component is the mechanism for removing pollutants from the land surface and subsurface and transporting them to a receiving water body.

The flows and pollutants from each point also are represented by hourly time series. The constituents and their concentrations relate to wastewater characteristics and the treatment process. Industrial point-source characteristics are different from municipal. The effluent from a secondary treatment plant is represented differently from effluent from a tertiary plant.

Chemical and biological reactions take place only in the receiving waters. These reactions are represented by first-order relationships such as:

$$H + NO_3 \xrightarrow{\text{Bacteria}} 1/2\ N_2 + 5/4\ O_2 + 1/2\ H_2O$$

This relationship describes a reduction of nitrates to nitrogen gas. The process, known as denitrification, results in the removal of nitrogen-related nutrients through the venting of nitrogen gas. The rate of denitrification is dependent upon dissolved oxygen concentrations and the level of thriving bacterial populations. The other biological and chemical relationships represented in the model are similar.

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The biological growth and death are based on Liebig's Law of the Minimum. Organisms grow in proportion to the least available nutrient or life requirement. The assumed requirements are phosphorus, nitrogen, carbon, light, and heat, each of which is represented in the model. When the limits of a requirement are reached, biological growth stops. When the available resources decline, organisms die.

In nature and in the model, the aquatic physical, chemical, and biological processes are interdependent. Dissolved oxygen, for example, is affected by BOD, temperature, and phytoplankton and zooplankton populations. In turn, dissolved oxygen affects denitrification. The availability of nitrogen and phosphorus affects phytoplankton and zooplankton. These organisms are influenced by stream temperature and light which, in turn, are determined by streamflow and depth. Figure 5 illustrates this interdependency as represented by QUALITY.

Summarizing, the sources of constituents represented in the model are pervious surface washoff, impervious surface washoff, groundwater, point-sources such as municipal and industrial wastewater treatment plants, and bottom sediments. The constituents that are represented are heat, dissolved oxygen, biological and chemical oxygen demand, ammonia, nitrate, phosphate, and chlorophyll a. The concentration of each constituent over a time interval is the algebraic sum of the mass contributions of each source and the losses due to physical, chemical, or biological transformations divided by the flow.

Toxicity Index

The toxicity index model was developed by Dr. Richard E. Sparks and associates at the Illinois Natural History Survey to demonstrate how existing water quality monitoring data could be used to evaluate the suitability of a lake or stream for fish life, and if the water was unsuitable, to determine which factors were responsible.

The bluegill sunfish, *Lepomis macrochirus*, was used as the reference organism because it is a panfish species which is common in North America and because its sensitivity to many chemicals has been determined in the laboratory. The toxicity units, therefore, were called bluegill toxicity units (BGTU). A BGTU value equal to 1.0 is lethal and would kill about 50% of the fish in 4 days. A value greater than 1.0 would kill most fish in a shorter period of time, and a value less than 1.0 is considered sublethal, although values close to 1.0 might kill a few sensitive fish over a period of days.

The water quality parameters were divided into three categories: limiting factors, modifying factors, and toxicants. The limiting factors are temperature, pH, and dissolved oxygen, which must be within a certain range to permit fish to survive. We included a wide range within which bluegills can survive for several days, as well as a narrower range within which bluegills can not only survive indefinitely, but also carry on normal functions such as growth and reproduction. Temperature, pH, and dissolved oxygen also are modifying factors, in that they modify the toxicity of some chemicals by changing the chemical equilibria in the water or the sensitivity of the fish.

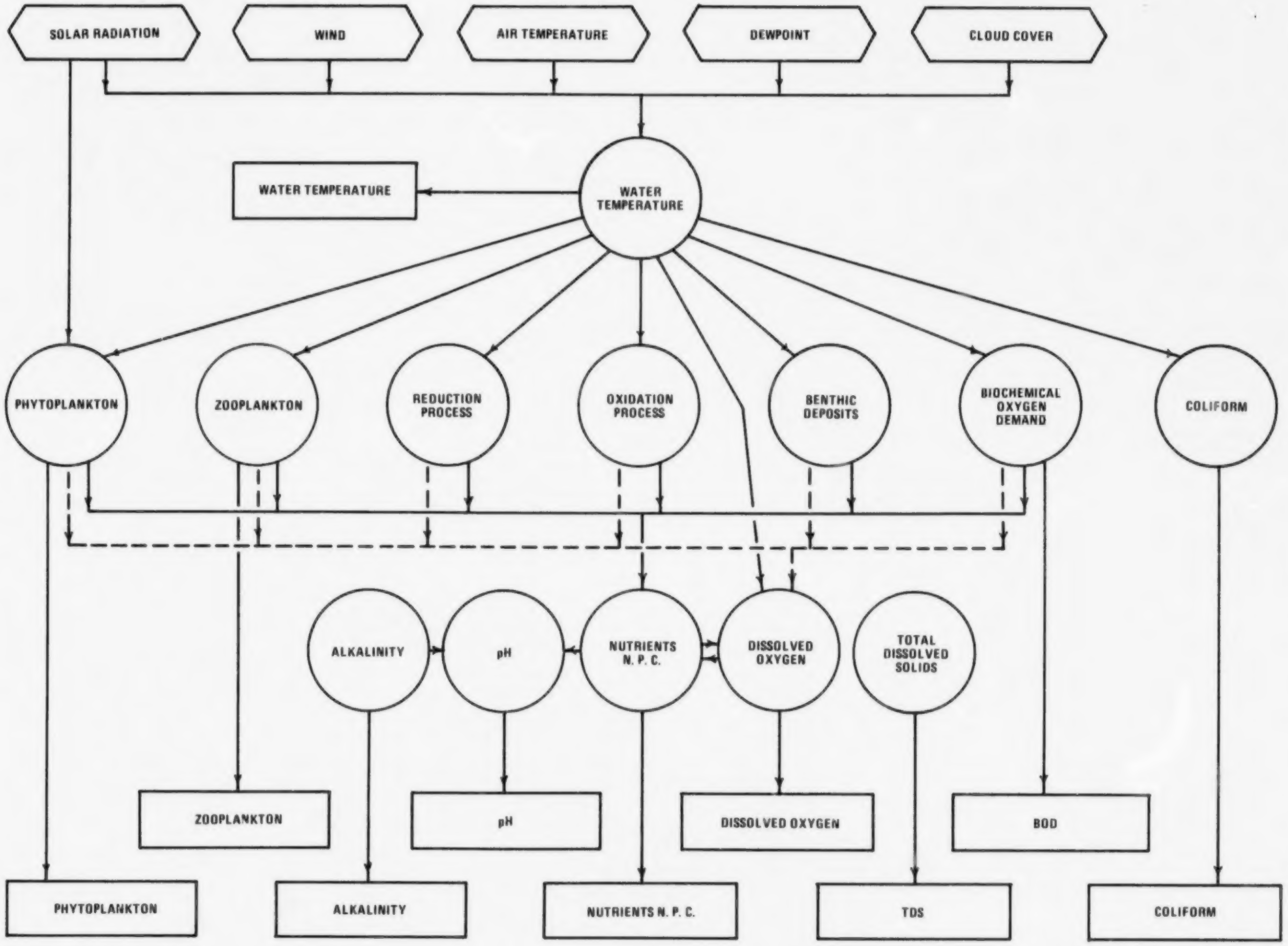
SOLAR RADIATION
WIND
AIR TEMPERATURE
DEWPOINT
CLOUD COVER
WATER TEMPERATURE
WATER TEMPERATURE
PHYTOPLANKTON
ZOOPLANKTON
REDUCTION PROCESS
OXIDATION PROCESS
BENTHIC DEPOSITS
BIOCHEMICAL OXYGEN DEMAND
COLIFORM
ALKALINITY
pH
NUTRIENTS N. P. C.
DISSOLVED OXYGEN
TOTAL DISSOLVED SOLIDS
ZOOPLANKTON
pH
DISSOLVED OXYGEN
BOD
PHYTOPLANKTON
ALKALINITY
NUTRIENTS N. P. C.
TDS
COLIFORM

Figure 5. Flow diagram for the QUALITY Model.

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Calcium also is a modifying factor because the greater the calcium concentration in water the less sensitive are bluegills and other fish to certain toxicants such as heavy metals. Twenty toxicants have been tested for toxic affects on bluegills and were used in computing toxicity indices.

The joint toxicity of all the chemicals present at a particular water quality sampling station at a particular sampling time was estimated by adding the toxicities contributed by the individual chemicals. This estimate of the joint toxicity is the toxicity index, while the toxicity contributed by any particular chemical is defined as a component toxicity.

In order to verify the assumption that the joint toxicity of a complex mixture can be estimated by adding up component toxicities, the toxicity of ammonia, LAS detergent, and zinc to bluegills was determined testing the chemical singly and in combination. The toxicity of the mixture as predicted by the toxicity index was then compared to the measured toxicity. The toxicity of the mixture was significantly underestimated by the toxicity index, indicating that the toxic effects of the components are more than additive. The importance of this finding is that water quality standards which consider each toxicant singly may not adequately protect aquatic organisms which are exposed to many toxicants simultaneously. Brown, Shurben, and Shaw (1970) also found that a toxicity index system they used underestimated the toxicity of severly polluted rivers, but they felt that, in view of the difficulties in making such an assessment of a river water, the relationship between predicted and observed values was sufficiently acceptable to have useful application.

Lloyd and Jordan (1964) found that a similar index system consistently underestimated the toxicity of sewage effluents and that the relation between the predicted and observed toxicity was described by the function:

$$y = 1.25x - 0.59$$

where y is the observed toxicity and x the predicted toxicity.

The effect of pH, temperature, hardness and dissolved oxygen levels on toxicity will be illustrated by several examples.

The highest total ammonia concentration in the Illinois River at Hardin was 4.50 mg liter^{-1} in 1973. The toxic un-ionized portion of the total ammonia concentration can be calculated using equations developed by Ball (1967). If the pH remains constant, but the water temperature varies between 5 and 30^{o}C, the following concentrations of un-ionized ammonia occur:

total ammonia = 4.50 mg liter^{-1}
(maximum concentration at Hardin, Illinois, 1973)

Temp. ^{o}C	pH	NH_3 (u)	BGTU
5	8	0.057	0.024
10	8	0.082	0.035
15	8	0.118	0.051
20	8	0.169	0.073
25	8	0.242	0.105
30	8	0.342	0.149

Thus, as water temperature increases the toxicity increases by a factor of six, due to the six-fold increase in concentration of un-ionized ammonia. The next table shows that when the temperature is constant at 20°C, but the pH varies within a range considered safe for fish, the un-ionized ammonia changes by a factor of 500, producing a 500-fold change in toxicity.

total ammonia = 4.50 mg liter^{-1}
(maximum concentration at Hardin, Illinois, 1973)

Temp. °C	pH	NH_3 (u)	BGTU
20	6	0.002	0.001
20	7	0.018	0.008
20	8	0.169	0.008
20	9	1.266	0.550

In neither example did the toxicity increase to a lethal level of 1.0, but with a total ammonia concentration of 4.50 mg liter^{-1}, a pH of 9, and a water temperature of 20°C, fish would be exposed to half the lethal level of un-ionized ammonia, a condition which undoubtedly would stress the fish.

The next example shows how the level of dissolved oxygen and hardness modify toxicity by modifying the susceptibility of fish to zinc. The maximum concentration of zinc in the Illinois River in 1972 was 0.2 mg liter^{-1} and occurred at Pekin. The table below shows that the toxicity of zinc would be reduced 5.5 times if the dissolved oxygen concentration remained constant at 6 mg liter^{-1} while the hardness of the water increased from 50 to 300 mg liter^{-1} (as $CaCO_3$).

zinc concentration = 0.2 mg liter^{-1}
(maximum concentration at Pekin, Illinois River, 1972)

Dissolved Oxygen mg liter^{-1}	Hardness mg liter^{-1} (as $CaCO_3$)	BGTU
6	50	0.082
6	100	0.044
6	200	0.023
6	300	0.015

It is believed that calcium, which is usually the major contributor to hardness, exerts a protective effect by reducing the permeability of fish gills to heavy metals. On the other hand, low oxygen levels stress fish, and this adds to the stress exerted by the toxicant. Thus, the fish's resistance is lowered, as shown below:

zinc concentration = 0.2 mg liter^{-1}
(maximum concentration at Pekin, Illinois River, 1972)

Dissolved Oxygen mg liter^{-1}	Hardness mg liter^{-1} (as $CaCO_3$)	BGTU
8	160	0.021
7	160	0.024
6	160	0.028
5	160	0.035
4	160	0.052
3	160	0.130

Note that the toxicity does not begin to change rapidly until the dissolved oxygen level drops below 4 mg liter^{-1}. When a dissolved oxygen level of 2 mg liter^{-1} was substituted in the equations, a very large value for toxicity was obtained, indicating that fish would be killed under these conditions. The hardness value of 160 mg liter^{-1} is typical for the Illinois River, and there have been places in the river and its backwaters where dissolved oxygen levels have been as low as 2 mg liter^{-1}. Of all the modifying factors, dissolved oxygen had the greatest impact upon the toxicity of chemicals to fishes.

The greatest deficiency of the toxicity index and similar indices is that they underestimate the toxicity of complex mixtures. If the indices consistently underestimate toxicities by a certain amount, the work of Lloyd and Jordan (1964) indicates that formulas could be developed for correcting the indices. Another deficiency is that these indices estimate lethal effects, whereas we would really like to know what levels of toxicants will permit organisms to thrive and perpetuate themselves indefinitely. Herbert, Jordan, and Lloyd (1965) felt that fish populations could maintain themselves in water where the total toxicity index was below 0.2 units. Brown, Shurben, and Shaw (1970) subsequently pointed out that the observed fish populations living in streams with index values close to 0.2 (range 0.22 to 0.40) may have been maintained by movement or recruitment from areas where the index was lower.

The toxicity index is useful in locating the places and times where conditions approach lethal levels for fish. It is also useful in determining which factors are contributing the most to the total toxicity at a given location. The toxicity index also provides a logical way of integrating information on environmental factors and the susceptibility of aquatic organisms.

The following section outlines in detail the method for calculating the toxicity index. The treatment is arranged into separate discussions of each parameter. Parameters are arranged alphabetically.

Ammonia—

Ammonia gas is soluble in water in the form of ammonium hydroxide to the extent of 100,000 mg liter^{-1} at 20°C. Ammonium hydroxide readily dissociates into ammonium and hydroxyl ions as follows:

$$NH_3 + H_2O = NH_4^+ + OH^-$$

The toxicity of ammonia is related only to the un-ionized portion. Most agencies, however, report total ammonia concentrations; therefore, before incorporating this into the toxicity index, the concentration of un-ionized ammonia must be determined. The un-ionized portion of the total ammonia concentration can be calculated by the following formula (Ball 1967):

$$NH^3(u) = \text{total ammonia} \times \frac{1}{1 + \text{antilog (pka} - \text{pH)}}$$

where pka = the negative logarithm of the ionization constant:

$$pka = -0.03229 \text{ (temp °C)} + 10.05333$$

Total dissolved solids concentration also influences the degree of dissociation, but this has not yet been incorporated into the formula. Also, many measurements of ammonia concentration must be converted from mg liter^{-1} as NH_3-N to mg liter^{-1} as NH_3 by multiplying by 1.2159.

Merkens and Downing (1957) found that the toxicity of un-ionized ammonia increased with decreased oxygen tension. This relationship can be estimated by the equation:

$$x/x_s = 0.013297 \text{ (D.O. \% saturation)} - 0.32965$$

where x = the 96-hour LC50 at the lowered dissolved oxygen concentration,
Note: LC50 represents a concentration likely to be lethal to 50% of the bluegills in 96 hours.
x_s = the 96-hour LC50 at 100% saturation

Unfortunately only two levels of dissolved oxygen were tested. The predictive equation was computed without statistical analysis assuming a linear relationship between x/x_s and dissolved oxygen concentration.

Multiplying x/x_s by the 96-hour LC50 of $NH_3(u)$, determined with dissolved oxygen at 100% saturation, gives the 96-hour LC50 modified for the given conditions of dissolved oxygen concentration.

The 96-hour LC50 of 2.3 mg liter^{-1} $NH_3(u)$ for bluegill (unpublished data) will be used in the calculation. This test was run under conditions of 87% saturation of dissolved oxygen. This value is adjusted to 1.9 mg liter^{-1} $NH_3(u)$ when corrected to account for test conditions with dissolved oxygen concentration at a 100% saturation.

Data on the effect of temperature on the lethal threshold are conflicting. Although data were not available for bluegill, Brown (1968) indicates that un-ionized ammonia is twice as toxic to rainbow trout at a temperature of 3°C than it is at a temperature of 10°C. Herbert (1962) found that above 10°C, temperature had no effect on the lethal threshold for rainbow trout. Until more data are available concerning the effect of temperature on the toxicity of ammonia, this modifying factor will not be incorporated into the index. Data from Downing and Merkens (1955) indicate that pH has no effect on the toxicity of $NH_3(u)$ to rainbow trout. No data could be found concerning the effect of water hardness on the toxicity of $NH_3(u)$.

The steps for calculating the component toxicity due to ammonia are as follows:

$$\text{dissolved oxygen \% saturation (DO\%)} = (\text{DOPPM} \div \text{DO100}) \times 100$$

$$\text{pka (PKA)} = (-0.03229 \times \text{TEMP}) + 10.05333$$

$$\text{unionized ammonia, mg liter}^{-1} \text{ (UAMM)} = \text{AMM} \div [1 + \text{antilog (PKA - PH)}]$$

$$\text{correction factor (CFDO)} = (0.013297 \times \text{DO\%}) - 0.32965$$

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corrected 96-hr LC50 (CLC50) = LC50 x CFDO
Note: if LC50 x CFDO $\leq$ 0, then CLC50 = 0.001
ammonia component toxicity (AMMTOX) = UAMM $\div$ CLC50

Where total ammonia concentration, mg liter^{-1}

water temperature, °C = TEMP
pH = PH
dissolved oxygen, mg liter^{-1} = DOPPM
dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100
96-hour LC50 at DO100, mg liter^{-1} (LC50) = 1.9

Arsenic--

Inorganic arsenic can occur in four oxidation states (+5, +3, 0, -3) under conditions in aquatic systems (Ferguson and Gavis 1972), as follows:

Valence	Compound
+5	H_3AsO_4
+5	$H_2AsO_4^-$
+5	$HAsO_4^{2-}$
+5	AsO_4^{3-}
+3	H_3AsO_3
+3	$H_2AsO_3^-$
+3	$HAsO_3^{2-}$
+3	$HAsS_2$
+3	As_2S_3
+3	AsS_2^-
0	As
-3	AsH_3

The sulfur-containing species are precipitates and would concentrate in the sediment. Thus, they would not affect fish in the water column. Elemental arsenic is not considered significant because it rarely occurs and AsH_3 only occurs under extremely low Eh conditions (less than -0.25 volts) which are not likely to occur in the water column. We are concerned, therefore, only with the +5 and most of the +3 forms of arsenic. Arsenate (+5) is the stable form in aerated water with a pH of 8 to 9, whereas arsenite (+3) is the stable form with little or no dissolved oxygen and with a pH of 6 to 7 (Ferguson and Gavis 1972).

In addition to the inorganic forms, certain fungi, yeasts, and bacteria are known to methlyate arsenic to the gaseous derivative arsine (Ferguson and Gavis 1972). These compounds are not incorporated into the index, since their toxicities have not been evaluated.

The toxicity of arsenic increases greatly when it is reduced from the arsenate to the arsenite. However, except for a few oxidation-reduction reactions that are used in analytical chemistry, very little information exists concerning the rates of arsenic reactions in solution (Ferguson and Gavis 1972). Thus, it is not possible to calculate accurately relative amounts of arsenate and arsenite in solution.

For the purposes of the toxicity index, the component toxicity of arsenic is computed assuming that all of the arsenic present is in the arsenite form. There are no data available on the effects of modifying factors upon the tolerance of fish to arsenic.

Gilderhus (1966) reports a 96-hour LC50 for bluegills of 35 mg liter^{-1} as sodium arsenite (20.2 mg liter^{-1} as arsenic). Although no data for the bluegill were available concerning arsenate toxicity, Sorensen (1976) reports a 48-hour LC50 for green sunfish of 150 mg liter^{-1} as arsenate.

The steps for calculating the component toxicity due to arsenite are as follows:

$$\text{arsenite component toxicity (AS3TOX)} = \text{AS} \div \text{LC50}$$

where total arsenic concentration, mg liter^{-1} = AS

arsenite 96-hour LC50, mg liter $^{-1}$ (LC50) = 20.2

Boron--

The only datum available on the toxicity of boron to bluegill is a 24-hour LC50 of 15,000 mg liter^{-1} as boron trifluoride (2,393 mg liter^{-1} as B) (Turnbull, DeMann, and Weston 1954). It is possible that this toxicity is attributable to the fluoride rather than the boron. A detailed study is needed to determine precisely boron toxicity to bluegill. However, until more data are available, the 24-hour LC50 value of 2,393 mg liter^{-1} as B may be used. No data are available concerning the effects of modifying factors upon boron toxicity.

The steps for calculating the component toxicity of boron are as follows:

$$\text{boron component toxicity (BTOX)} = \text{B} \div \text{LC50}$$

where boron concentration, mg liter^{-1} = B

boron 24-hour LC50, mg liter^{-1} = LC50

Cadmium--

Acute bioassays for cadmium were conducted on bluegills by Pickering and Henderson (1966) and Eaton (1974). Pickering and Henderson computed a 96-hour LC50 of 1.94 mg liter^{-1} as Cd with the test water having a hardness of 20 mg liter^{-1} as CaCO$_3$. Eaton conducted two tests with a water hardness of 207 mg liter^{-1} as CaCO$_3$ and computed lethal thresholds of 17.2 and 24.2 mg liter^{-1} as Cd. Dissolved oxygen, water temperature, and pH were approximately the same in all of these tests. Thus, the toxicity of cadmium to bluegills apparently varies with water hardness. A crude predictive equation defining the effect of hardness upon the 96-hour LC50 of cadmium, based upon the results of the studies cited above, is incorporated into the calculation of the component toxicity due to cadmium. Unfortunately, test data are available for only two water hardnesses. The predictive equation used here assumes a linear relationship between water hardness and the 96-hour LC50 without statistical analysis. Improvements

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could be made on this equation if more data were available.

There is no information available concerning the effects of other modifying factors upon the tolerance of bluegill to cadmium. Also, there is no information available concerning the possible variation in cadmium toxicity due to effects of modifying factors upon the chemical form and solubility of cadmium.

The steps for calculating the component toxicity due to cadmium are as follows:

96-hr LC50, mg liter^{-1} as Cd, corrected for hardness effects
(CLC50) = (0.100321 x HARD) - 0.066417
Note: if CLC50 $\leq$ 0, then CLC50 = 0.001
cadmium component toxicity (CDTOX) = CD $\div$ CLC50

where cadmium concentration, mg liter^{-1} as Cd = CD
hardness, mg liter^{-1} as $CaCO_3$ = HARD

Chromium, Hexavalent--

Acute bioassays for hexavalent chromium were conducted on bluegills in both soft water (total hardness = 20 mg liter^{-1} as $CaCO_3$) and hard water (total hardness = 360 mg liter^{-1} as $CaCO_3$) (Pickering and Henderson 1966). The 96-hour LC50 for the soft-water test was 118 mg liter^{-1} as Cr^{+6} and for the hard-water test, 133 mg liter^{-1} as Cr^{+6}. This difference indicates a slight effect of hardness on hexavalent-chromium toxicity. The hardness effect has not been incorporated into the calculations for component toxicity due to hexavalent chromium.

Cairns and Scheier (1959) found that temperature did not have a significant effect upon the median concentration of hexavalent chromium toxic to bluegills with tests run at 18 and 30°C.

There is no information concerning the effects of pH and dissolved oxygen on the tolerance of fish to hexavalent chromium. Also, there is no information concerning effects of modifying factors on the chemical forms or solubility of chromium.

For purposes of the toxicity index the mean of the soft and hard water 96-hour LC50's reported by Pickering and Henderson (1966) will be used. This mean is 125.5 mg liter^{-1} as Cr.

hexavalent chromium component toxicity (CR6TOX) = CR6 $\div$ LC50

where hexavalent chromium concentration, mg liter^{-1} = CR6

hexavalent chromium 96-hr LC50, mg liter^{-1} (LC50) = 125.5

Chromium, Trivalent--

The data presented in McKee and Wolf (1963) indicate that a conclusion cannot be drawn concerning differences in the relative toxicities of hexavalent chromium and trivalent chromium. On this basis the 96-hour LC50 of 125.5 mg liter^{-1} as Cr, based on the hexavalent toxicity, also will be used for the trivalent-chromium component toxicity calculations. Bioassays have not been conducted to determine the toxicity of trivalent chromium to bluegills or to determine the effects of modifying factors.

The steps for calculating the component toxicity due to trivalent chromium are as follows:

$$\text{trivalent chromium component toxicity (CR3TOX)} = \text{CR3} \div \text{LC50}$$

where trivalent chromium concentration, mg liter^{-1} = CR3
trivalent chromium 96-hr LC50, mg liter^{-1} = 125.5

Copper—

A review of the literature indicates that dissolved oxygen and hardness affect the toxicity of copper to aquatic organisms. There are no data, however, pertaining to the effects of dissolved oxygen concentration upon the toxicity of copper to the bluegill. Lloyd (1961) obtained data on dissolved oxygen effects using rainbow trout, as follows:

x_s/x	D.O. (% saturation)
1.00	100
1.05	82
1.1	71
1.2	58
1.3	48
1.4	40
1.5	33

where x_s = concentration of copper at 100% dissolved oxygen saturation
x = equitoxic concentration at a lower value of dissolved oxygen

For use in the predictive equation used here, Lloyd's values were modified as follows:

x/x_s	log D.O. (% saturation)
1.000	2.000
0.952	1.914
0.909	1.851
0.833	1.763
0.769	1.681
0.714	1.602
0.667	1.519

The reciprocal of x_s/x was used in order that the resultant calculation could be used as multiplication factor. The log of the percent saturation of dissolved oxygen was used because it resulted in better correlation between the correction factor and dissolved oxygen concentration. The relationship between the correction factor and dissolved oxygen concentration can be represented:

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$$x/x_s = [0.72210 \times (\log \text{D.O. \% saturation})] - 0.43707$$

The correction factor x/x_s is used to adjust copper toxicity to the given dissolved oxygen conditions.

Several sources provide data useful for computing a predictive equation for the 96-hour LC50 for copper as modified by hardness:

D.O. % saturation	Hardness mg liter $^{-1}$ as $CaCO_3$	96-hour LC50 mg liter^{-1} Cu^{++}	96-hour LC50 mg liter^{-1} Cu^{++} corrected to D.O.=100% saturation	Source
72	20	0.66	0.73	Pickering and Henderson (1966)
72	360	10.2	11.24	Pickering and Henderson (1966)
76	45	1.1	1.19	Benoit (1975)
65.4	46	0.74	0.84	Trama (1956b)

Percent saturation was computed based on the concentration and water temperature reported in the literature and the altitude of the laboratory. Since the dissolved oxygen levels during the bioassays were below saturation, the 96-hour LC50s obtained by these investigators had to be corrected to 100% saturation, using the correction factor x/x_s described above. The corrected data were used to determine the following regression equation (significant at $p = 0.01$):

$$\text{96-hour LC50 mg liter}^{-1}\ \text{Cu} = [0.031834 \times (\text{hardness, mg liter}^{-1}\ \text{as}\ CaCO_3)] - 0.248404$$

Copper toxicity appears to be related to both soluble Cu^{++} and $CuCO_3$ concentrations (Shaw and Brown 1974). This indicates that factors affecting the chemical forms and solubility of copper will not affect significantly copper toxicity. Data in the literature also show that pH has no effect on the toxicity of copper to fish (Shaw and Brown 1974). Cairns, Heath, and Parker (1975) suspected that temperature had an effect on the lethal toxicity of copper. Quantitative data are lacking, and for the present, it is best to consider that temperature has no effect.

The steps for calculating the component toxicity due to copper are as follows:

dissolved oxygen % saturation (DO%) = (DOPPM ÷ DO100) x 100
96-hr LC50, mg liter^{-1} as Cu, corrected for hardness effect, DO = 100% saturation (LC50) = [0.031834 x (HARD)] - 0.248404
correction factor for dissolved oxygen (CFDO) = [0.72210 x log (DO%)] - 0.43707

96-hr LC50, corrected for hardness and dissolved oxygen effects (CLC50) = LC50 x CFDO
Note: if CLC50 $\leq$ 0, then CLC50 = 0.001

copper component toxicity (CUTOX) = CU ÷ CLC50

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where copper concentration, mg liter^{-1} = CU
hardness, mg liter^{-1} as $CaCO_3$ = HARD
dissolved oxygen, mg liter^{-1} = DOPPM
dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100

Cyanide--

In aqueous solutions of cyanides, the cyanide group CN can exist in different forms, including ionized cyanide (CN^-) and molecular cyanide (HCN). In addition, cyanide can be bound up in complexes with heavy metals and other compounds (Doudoroff 1976). Data from Doudoroff, Leduc, and Schneider (1966) demonstrated that the acute toxicity of solutions containing complex metal cyanides was related only to the molecular HCN concentration. An exception seemed to be silver-cyanide complex with symptoms indicating heavy metal poisoning which was considered the result of the toxicity of the complex anions or of silver cations.

It may not be feasible, even when the general chemical composition and pH of a complex cyanide solution such as a polluted stream are known, to reliably compute the molecular cyanide level (Doudoroff 1966). This is reasonable if there is only free cyanide present in the stream, since at pH's normally encountered in streams of Illinois, molecular cyanide will comprise over 85% of the total cyanide. However, problems arise when the nontoxic cyanide complexes are present. The cyanide in these complexes is measured as part of the total cyanide concentration. Therefore, considering the total cyanide concentration equivalent to molecular cyanide might overestimate the cyanide component toxicity.

The literature indicates that temperature, pH, and dissolved oxygen affect the acute toxicity of cyanide as measured by the 96-hour LC50. Cairns and Scheier (1963) tested bluegill in a static system and determined the 96-hour LC50's in soft (50 mg liter^{-1} $CaCO_3$ hardness) and hard (180 mg liter^{-1} $CaCO_3$ hardness) water at 18 and 30°C. The results are as follows:

Temp, °C	Hardness, mg liter^{-1} as $CaCO_3$	96 hour LC50, mg liter^{-1} as CN
18	50	0.18
18	180	0.17
30	50	0.13
30	180	0.14

The data show that the levels of hardness tested did not affect the toxicity of cyanide. A predictive equation was determined using the temperature and 96-hour LC50 data:

$$\text{96-hour LC50} = -0.003333 \times (\text{temp }^\circ\text{C}) + 0.235$$

There are problems associated with using these data. Doudoroff (1976) states that there is a fairly rapid loss of cyanide from static test solutions. As a result the above values might be slightly high. In addition, the LC50's were based on total KCN concentrations rather than molecular cyanide. However, this is the only experiment that tested cyanide toxicity at more than

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one temperature. The predictive equation was determined assuming a linear relationship between the two temperatures and the 96-hour LC50's without statistical analysis.

The 96-hour LC50's of free cyanide and molecular cyanide differ little in the pH range 6.8 to 8.3. Beyond this (to pH 9.3) values diverged markedly, with the free cyanide LC50's increasing and the HCN LC50's decreasing. Increased apparent toxicity of molecular cyanide with elevated test pH is believed to result from the CN^- anions penetrating the gill epithelium, though less readily than the molecular forms. Thus, the toxicities of these solutions are enhanced as the pH increases.

Data used in this modification are derived from tests 1 and 2 of Broderius, Smith, and Lind (1977). These data were used for most calculations in that paper. Thus, at 20°C and pH 7.22, the 96-hr LC50 for free cyanide was found to be 0.12023 mg liter^{-1}. The ratio between the molecular toxicity and ionic toxicity was found to be 2.3 (fairly constant). K_{HCN} at 20°C = 4.7000×10^{-10}.

Assuming that the slope coefficient of the Cairns and Scheier (1963) equation above is valid, the intercept is changed to fit the data of Broderius, Smith, and Lind (1977), as follows:

$$\text{LC50} = -0.00333 \times (\text{TEMP}) + 0.18689$$

Equation 4 of Broderius, Smith, and Lind (1977) is then used to correct this value for the pH effect. Equation 4 reduced to:

$$\text{LC50}' = \frac{2.307794 + \dfrac{1.0846634 \times 10^{-9}}{\text{antilog pH}}}{2.317927 + \dfrac{4.7366335 \times 10^{-10}}{\text{antilog pH}}}$$

This value (LC50') is then corrected for dissolved oxygen to give the CLC50.

There is limited information concerning the effects of dissolved oxygen on the toxicity of cyanide. Cairns and Scheier (1958a) investigated the effects of low dissolved oxygen on the toxicity of cyanide to bluegill. However, the dissolved oxygen level was only reduced 2 hours daily. Therefore, the results from this test were not adequate for use in the toxicity index. Burdick, Dean, and Harris (1958) conducted experiments with smallmouth bass to determine effects of low oxygen on cyanide toxicity. Their results were as follows:

Dissolved oxygen (% saturation)	Threshold concentration, mg liter^{-1} as CN
100	0.104
44	0.086

Percent saturation of dissolved oxygen was computed based on the dissolved oxygen concentrations, altitude of the laboratory and test temperature.

For use in the toxicity index the data were modified as follows:

Dissolved oxygen (% saturation)	x/x_s
100	1.00
44	1.21

where x = threshold concentration at 100% dissolved oxygen saturation
x_s = threshold concentration at the lower dissolved oxygen

This relationship can be represented

$$x/x_s = 0.003089 \text{ (D.O. \% saturation)} + 0.691071$$

Multiplying x/x_s times the 96-hour LC50' of cyanide determined at 100% saturation gives a 96-hour LC50 corrected for the lower dissolved oxygen condition. Unfortunately, only two levels of dissolved oxygen were tested. The predictive equation was calculated as a linear relationship between x/x_s and dissolved oxygen concentrations without statistical analysis.

The steps for calculating the component toxicity of cyanide are as follows:

dissolved oxygen, % saturation (DO%) = (DOPPM ÷ DO100) x 100

96-hr LC50, mg liter^{-1} as CN, modified for temperature (LC50) = (-0.00333 x TEMP) + 0.18689

corrected LC50 accounting for temperature and pH effects (LC50') =

$$\left(\frac{2.307794 + \frac{1.0846634 \times 10^{-9}}{\text{antilog PH}}}{2.317927 + \frac{4.7366335 \times 10^{-10}}{\text{antilog PH}}} \right) \times \text{LC50}$$

correction factor for dissolved oxygen (CFDO) = (0.003089 x DO%) + 0.691071

corrected LC50 accounting for temperature, pH, and dissolved oxgyen effects (CL50) = LC50' x CFDO
Note: if LC50' x CFDO $\leq$ 0, then CL50 = 0.001

cyanide component toxicity (CNTOX) = CN ÷ CLC50

where cyanide concentration, mg liter^{-1} = CN
water temperature, °C = TEMP
dissolved oxygen, mg liter^{-1} = DOPPM
dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100
pH = PH

Dissolved Oxygen--

Moore (1942) reported that 3.1 mg liter^{-1} was the highest observed minimum oxygen concentration which killed bluegill within 24 hours at summer temperatures and 0.8 mg liter^{-1} was the highest observed minimum dissolved oxygen concentration which killed bluegill within 48 hours at winter temperatures. These values are used in the toxicity index, with summer months representing the period March through November and winter months representing the period December through February.

The effects of low dissolved oxygen on the growth and reproduction of fish also is incorporated in the toxicity index. A minimum dissolved oxygen concentration of 5.0 mg liter^{-1} is recommended to maintain good fish populations (U.S. Environmental Protection Agency 1976).

If the dissolved oxygen drops below these minimum values, the toxicity index calculations account for conditions lethal to bluegills or conditions that are detrimental for the growth and reproduction of bluegills.

Fluoride--

No reports on the toxicity of fluoride to bluegills were found. However, a mean 96-hour LC50 between those reported for fluoride to rainbow trout and carp was assumed applicable, as suggested by Lubinski (1975). Neuhold and Sigler (1960) reported that the LC50 of fluoride to rainbow trout was between 2.7 and 4.7 mg liter^{-1}, while the fluoride LC50 to carp was between 75 and 95 mg liter^{-1}. The mean value is 44.4 mg liter^{-1} and this is the 96-hour LC50 incorporated into the toxicity index. No information exists regarding the effects of modifying factors upon the toxicity of fluoride to fishes.

The steps for calculating the component toxicity due to fluoride are as follows:

$$\text{fluoride component toxicity (FLTOX)} = \text{FL} \div \text{LC50}$$

where fluoride concentration, mg liter^{-1} = FL
fluoride 96-hr LC50, mg liter^{-1} = LC50

Hardness--

There are no data available which indicate that hardness normally encountered in freshwater systems is directly lethal to bluegills. Hardness, however, does indirectly affect the toxicity of a number of substances to bluegills. Thus, it appears as a modifying factor in many of the calculations of component toxicity.

Hydrogen Ion--

Trama (1954a) reports the minimum and maximum tolerance limits to pH for bluegill as 4.0 and 10.4. Cairns and Scheier (1958b) report the minimum and maximum for bluegill as 3.6 and 10.5. The means of the upper and lower limits, 3.8 and 10.4, are used in the toxicity index. If the pH exceeds these values, the toxicity index calculations account for conditions lethal to bluegills.

The chronic effects of pH also are incorporated into the toxicity index. Although there were no data on the chronic effects of pH on bluegill, the U.S. Environmental Protection Agency (1976) gives a range of 6.5 to 9.0 as safe for freshwater aquatic life. This range is based upon a number of chronic tests. The toxicity index calculations also account for conditions having chronic effects upon bluegills. Hydrogen ion concentration is an important modifying factor because it affects the tolerance of organisms to that substance. Thus, pH is incorporated into many of the component toxicity calculations.

Iron--

Doudoroff and Katz (1953) indicate that the available data on iron toxicity to fish do not provide many definite answers. Iron compounds can have pronounced effects on the pH and dissolved oxygen concentration of receiving waters. Thus, the toxicity of water containing iron compounds may be related to pH or dissolved oxygen rather than iron. This supports the earlier observations of Jones (1939), who found that the toxicity of solutions of both ferric chloride and ferrous sulfate in very soft water was due to their acidity. He reported that solutions with tolerable pH values were not harmful. Ellis, (1937), however, found that 100 mg liter^{-1} of ferrous sulfate (32.9 mg liter^{-1} as Fe) at nontoxic pH levels was fatal to bluegill sunfish in 2.5 to 7 days. A 96-hour LC50 or lethal threshold was not given. Until more data are available these data will be used in the toxicity index.

Sanborn (1945) noted that an accumulation of ferric hydroxide on the gills appeared to be the cause of death of fish in ferrous sulfate solutions. This was caused by the oxidation at the gill surface of the soluble ferrous ion to the ferric form which quickly precipitates as ferric hydroxide. Considering this, total iron concentration will be used in the toxicity index calculations, since both the dissolved ferrous forms and the suspended ferric forms are factors in iron toxicity.

No data could be obtained concerning the effects of modifying factors upon the toxicity of iron.

The steps for calculating the component toxicity due to iron are as follows:

$$\text{iron component toxicity (FETOX)} = \text{FETOT} \div \text{LC50}$$

where total iron concentration, mg liter^{-1} = FETOT
lethal concentration of iron, mg liter^{-1} (LC50) = 32.9

LAS--

Linear alkylsulfonates [LAS] largely replaced alkyl benzyl sulfonates [ABS] as the active ingredient in detergents in 1965 (Sullivan and Swisher 1969). The methylene blue method is the most widely used technique for the quantitative measurement of anionic surfactants, however, and this technique does not differentiate between LAS, ABS, or other methylene-blue-chloroform-extractable substances [MBAS]. Sullivan and Swisher reported that LAS comprised 10 to 20% of the active MBAS.

Hokanson and Smith (1971) reported that temperature had no effect upon the lethal threshold of LAS to bluegill. They found that the toxicity increased with increased hardness. In contrast, McKim, Arthur, and Thorslund (1975) reported that hardness had little effect upon the 30-day toxicity of

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LAS to larval fathead minnows. Until more data are available, it is not possible to define the relationship between hardness and LAS toxicity. The effects of pH upon the toxicity of LAS to fishes has not been investigated.

Hokanson and Smith (1971) found that dissolved oxygen concentration affected the toxicity of LAS to bluegill. Their data, reproduced below, were useful in computing a predictive equation for the modifying effects of dissolved oxygen.

D.O. % saturation	Lethal Threshold mg liter^{-1}
91	2.2
89	2.1
58	1.9
35	0.5
25	0.4
24	0.2

The correlation coefficient between these variables was found to be 0.954, significant at $p = 0.01$. Regression analysis yielded the following relationship.

$$\text{lethal threshold for LAS (mg liter}^{-1}\text{)} = (0.029256 \times \text{DO\% saturation}) - 0.353387$$

The steps for calculating the component toxicity due to LAS are as follows:

$$\text{dissolved oxygen \% saturation (DO\%)} = (\text{DOPPM} \div \text{DO100}) \times 100$$

$$\text{96-hour LC50, corrected for dissolved oxygen effects (CLC50)} = (0.029256 \times \text{DO\%}) - 0.353387$$

Note: if $\text{CLC50} \leq 0$, then $\text{CLC50} = 0.001$

$$\text{corrected MBAS concentration (LAS)} = \text{MBAS} \times 0.15$$
$$\text{LAS component toxicity (LASTO)} = \text{LAS} \div \text{CLC50}$$

where MBAS concentration, mg liter^{-1} = MBAS
dissolved oxygen, mg liter^{-1} = DOPPM
dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100

Lead--

Lead is toxic to aquatic organisms and its toxicity is known to be affected by dissolved oxygen concentration and hardness. There are no data, however, pertaining to the effects of dissolved oxygen concentration upon the toxicity of lead to bluegills. Lloyd (1961) obtained data on dissolved oxygen effects using rainbow trout and these data were described in detail above in the discussion of copper. The same relationship is applicable to lead, as follows

$$x/x_s = [0.72210(\log \text{D.O.\% saturation})] - 0.43707$$

where x_s is the concentration of lead at 100% of the saturation concentration of dissolved oxygen and x is the equitoxic concentration at a lower concentration of dissolved oxygen. The correction x/x_s is used to adjust lead toxicity to the given dissolved oxygen conditions.

The toxicity of lead to fishes is affected by hardness, but the nature of this effect has not been investigated thoroughly. Pickering and Henderson (1966) determined 96-hr LC50's of 23.8 (26.2) mg liter^{-1} and 442.0 (486.9) mg liter^{-1} lead at hardnesses of 20 and 360 mg liter^{-1} as $CaCO_3$, respectively. Their work was conducted at a dissolved oxygen concentration of 72% saturation. The numbers in parentheses are 96-hr LC50's corrected to dissolved oxygen at 100% saturation. A crude regression equation was computed from these values, assuming a linear relationship between the two variables, as follows:

$$\text{96-hr LC50 (mg liter}^{-1}\text{ Pb)} = 1.355 \times \text{hardness (mg liter}^{-1}\text{ as CaCO}_3\text{)} - 0.9$$

It is likely that actual 96-hr LC50's will be lower than those generated by the above equation. Pickering and Henderson (1966) reported amounts of lead added to the test water rather than resulting toxicant concentrations. Since they noted the formation of precipiates upon the addition of the lead salts, the reported values probably are higher than actual concentrations in solution.

There were no data available concerning the effects of other modifying factors upon the tolerance of fishes to lead. There also are no data available concerning possible variation of toxicity due to the effects of modifying factors upon the chemical form and solubility of lead.

The steps for calculating the component toxicity due to lead are as follows:

dissolved oxygen % saturation (DO%) = (DOPPM ÷ DO100) x 100
log of dissolved oxygen % saturation (LOG DO%) = log (DO%)

correction factor for dissolved oxygen effect (CFDO) =
0.72210 (LOG DO%) - 0.43707

96-hr LC50, mg liter^{-1}, at D.O. adjusted to 100% saturation,
corrected for hardness effects (LC50) = 1.355(HARD) - 0.9

corrected 96-hr LC50, accounting for D.O. and hardness effects
(CLC50) = LC50 x CFDO

Note: if CLC50 $\leq$ 0, then CLC50 = 0.001

lead component toxicity (PBTOX) = PB ÷ CLC50

where lead concentration, mg liter^{-1} as Pb = PB
hardness, mg liter^{-1} as $CaCO_3$ = HARD
dissolved oxygen, mg liter^{-1} = DOPPM

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dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100

Manganese--

No reports of the toxicity of manganese to bluegills were found. Jones (1939) gave the lethal concentration for the stickleback, *Gasterosteus aculeatus*, as 40 mg liter^{-1}. This value will be used in the toxicity index until data are available for bluegill.

No data on the effects of modifying factors on manganese toxicity are available.

The steps for calculating the component toxicity due to manganese are as follows:

manganese component toxicity (MNTOX) = MN ÷ LC50

where manganese concentration, mg liter^{-1} = MN
manganese 96-hr LC50, mg liter^{-1} (LC50) = 40.

Mercury--

Mercury has three oxidation states: zero (elemental mercury), monovalent (mercurous componds, +1), and divalent (mercuric compounds, +2). Regardless of the mercury form present, the major portion of the mercury ultimately will reside in the sediments where, through microbial action, mono- and dimethyl mercury can be formed (U.S. Environmental Protection Agency 1976). All forms of methyl mercury are highly toxic to fish.

No data could be obtained concerning the acute toxicity of methyl mercury to bluegill. Most of the research on mercury has been restricted to investigation of chronic effects. However, Willford (1966) reported on the acute toxicity of phenylmercuric acetate to bluegill. Although a 96-hour LC50 was not reported, the 48-hour LC50 was 16.0 mg liter^{-1} (9.54 mg liter $^{-1}$ as Hg). Until more data are available, this 48-hour LC50 will be used in the toxicity index.

Kabata (1971) reported that water hardness did not affect the toxicity of mercury. No data could be found concerning the effects of other modifying factors upon the tolerance of fish to acute concentrations of mercury. In addition, there was no information available concerning possible variation of the acute toxicity of mercury due to modifying factors affecting its chemical form or solubility.

The steps for calculating the component toxicity due to mercury are as follows:

mercury component toxicity (HGTOX) = HG ÷LC50

where mercury concentration, ug liter^{-1} = HG
mercury 48-hour LC50, ug liter^{-1} as Hg (LC50) = 9540

Nickel--

Pickering and Henderson (1966) investigated the toxicity of nickel to bluegill. They determined the 96-hour LC50's at two different hardnesses (2.7 mg $liter^{-1}$ Ni at hardness of 20 mg $liter^{-1}$ as $CaCO_3$ and 39.6 mg $liter^{-1}$ Ni at hardness of 360 mg $liter^{-1}$ as $CaCO_3$). A crude regression equation was computed from these values, assuming a linear relationship between the two variables, as follows:

$$\text{96-hr LC50 (mg liter}^{-1}\text{ Ni)} = 0.100971 \text{ hardness (mg liter}^{-1}\text{ as } CaCO_3) + 3.250588$$

This equation needs to be refined as additional data become available.

Rehwoldt _et al._ (1972) found that temperature had no effect upon the lethal threshold concentration of nickel to six species of fish. The effects of other modifying factors on the tolerance of fish to nickel or on the chemical form and solubility of nickel have not been investigated.

The steps for calculating the component toxicity due to nickel are as follows:

$$\text{96-hr LC50, corrected for hardness effects (CLC50)} = 0.100971(\text{HARD}) + 3.250588$$

$$\text{nickel component toxicity (NITOX)} = \text{NI} \div \text{CLC50}$$

where nickel concentration, mg $liter^{-1}$ = NI
hardness, mg $liter^{-1}$ as $CaCO_3$ = HARD

Nitrate and Nitrite--

The only information regarding the toxicity of nitrate to bluegill is the single 96-hour LC-50 of 8,753 mg $liter^{-1}$ as NO_3 reported by Trama (1954c). Nitrite toxicity to bluegill has not been investigated.

Information on the relative toxicities of nitrate and nitrite to rainbow trout shows that nitrite is nearly 5,000 times more toxic than nitrate. Russo, Smith, and Thurston (1974) reported 96-hour LC50 for nitrite of 0.29 mg $liter^{-1}$ as N while Westin (1974) reported a 96-hour LC50 for nitrate as 1,360 mg $liter^{-1}$ as N.

McKee and Wolf (1963) stated that because nitrites are oxidized quickly to nitrates, they are seldom present in surface waters in significant concentrations. Russo, Smith, and Thurston (1974), however, noted that the amount of nitrite discharged by a wastewater treatment plant utilizing a nitrification process may result in nitrite concentrations that significantly affect the stream biota. In streams receiving effluents high in ammonia concentration, it is probable that nitrite could reach toxic levels if the dissolved oxygen concentration was low.

For the present, the nitrate and nitrite concentrations will be considered only as nitrate in the toxicity index. The steps for calculating the component toxicity due to nitrate and nitrite are as follows:

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conversion of nitrate + nitrite, mg liter^{-1} as N, to mg liter^{-1} as NO_3 (NO3) = NO3N x 4.4268

nitrate + nitrite component toxicity (NO3TOX) = NO3 ÷ LC50

where nitrate + nitrite concentration, mg liter^{-1} as N = NO3N

nitrate 96-hour LC50, mg liter^{-1} as NO_3 = 8,753

Phenol--

A review of the literature indicates that dissolved oxygen affects the toxicity of phenol to aquatic organisms (Lloyd 1961). The values used in determining the predictive equations for the effect of dissolved oxygen concentration are the same as those given in the discussion for copper. The relationship was represented by the equation

$$x/x_s = [0.72210 \times (\log \text{D.O. \% saturation})] - 0.43707$$

where x_s = concentration of phenol at 100% dissolved oxygen saturation

x = equitoxic concentration at a lower value of dissolved oxygen.

The correction factor x/x_s is used to adjust the phenol toxicity to the given dissolved oxygen concentration.

The 96-hour LC50 that is used in the calculations was determined by Trama (1955). He reported a value of 19.3 mg liter^{-1} phenol at a mean dissolved oxygen concentration of 67% saturation. When corrected to 100% saturation of dissolved oxygen, the 96-hour LC50 becomes 17.1 mg liter^{-1} phenol.

Hardness, temperature, and pH were found not to affect the lethal threshold concentration of phenol (Herbert 1962). No data were available regarding the possible variation in toxicity due to the effects that the modifying factors might have on the chemical form and/or solubility of phenol.

The steps for calculating the component toxicity due to phenol are as follows:

dissolved oxygen % saturation (DO%) = (DOPPM ÷ DO100) x 100

correction factor for dissolved oxygen (CFDO) = [0.72210 x log (DO%)] - 0.43707

96-hour LC50, corrected for dissolved oxygen effects (CLC50) = LC50 X CFDO

Note: if CLC50 $\leq$ 0, then CLC50 = 0.001

phenol component toxicity (PNTOX) = PHENOL ÷ CLC50

where phenol concentration, mg liter^{-1} = PHENOL

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dissolved oxygen, mg liter^{-1} = DOPPM

dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson 1957, Table 74) = DO100

phenol 96-hour LC50, mg liter^{-1} = 17.1

Residual Chlorine--

Residual chlorine is not a natural constituent of water. It is present intermittently as a result of point discharges. The most common point discharges of residual chlorine result from the use of chlorine to disinfect domestic wastewater, to control the growth of organisms in cooling water systems, or to bleach industrial products.

The very characteristics that make chlorine an excellent disinfectant also make it toxic to aquatic life. The toxicity of residual chlorine to aquatic life is dependent upon the total residual chlorine plus both free chlorine and the chloramines. Merkins (1958) found that the toxicity of free chlorine was slightly greater than that of chloramine, but that the difference was small enough not to warrant differentiation.

Snoeyink and Markus (1974) reported chlorine residuals for 24 wastewater treatment plants in central Illinois. Their samples showed an average value of 2.22 mg liter^{-1} with a standard deviation of 1.24 mg liter^{-1}. As chlorine usually is added to effluent at a constant rate and the effluent flow varies, residual chlorine at a given outfall should vary inversely with flow. Snoeyink and Markus (1974), however, found no apparent correlation between mean treatment plant flow and residual chlorine concentration.

Insufficient data are available regarding residual chlorine concentrations in the DuPage River basin for a rigorous model to be prepared. Rather, an average residual discharge concentration of 2.22 mg liter^{-1} will be assumed for all wastewater treatment plants in the basin. Residual chlorine will be discharged as a mass at a constant rate. The discharge mass will equal 2.22 mg liter^{-1} times the mean annual flow at the plant.

A decay rate for residual chlorine was estimated based upon data reported by Lewis and Heidinger (1978). The model assumes first order decay. The reader is directed to Lewis and Heidinger for further discussion.

Silver--

The acute toxicity of silver to bluegill has not been investigated. The stickleback, however, has been studied, with Jones (1939) and Anderson (1948) reporting similar values: 0.003 mg liter^{-1} and 0.0048 mg liter^{-1}, respectively. The mean of these values, 0.0039 mg liter^{-1}, will be used in the toxicity index until data for bluegill become available.

The steps for calculating the component toxicity due to silver are as follows:

silver component toxicity (AGTOX) = AG ÷ LC50

where silver concentration, mg liter^{-1} as Ag = AG
mean silver lethal threshold, mg liter^{-1} (LC50) = 0.0039

Temperature--

McKee and Wolf (1963) give two median tolerance limits of high temperature for bluegill. The two values are 35.0°C and 33.8°C, with a mean of 34.4°C. This mean value will be used in the toxicity index. If temperature equals or exceeds 34.4°C, the toxicity index calculations will determine that conditions lethal to bluegill exist. The tolerance limits of low temperature for bluegill have not been investigated.

Zinc--

A review of the literature indicated that the toxicity of zinc compounds to aquatic organisms is modified by both hardness and dissolved oxygen concentration. Several sources provide data useful for computing a predictive equation for the 96-hour LC50 for zinc:

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D.O. % saturation	Hardness mg liter $^{-1}$ as $CaCO_3$	96-hour LC50 mg liter^{-1} Zinc	Source
22.0	370	7.40	Pickering (1968)
39.0	370	10.60	Pickering (1968)
68.3	370	11.40	Pickering (1968)
89.0	46	3.32	Cairns and Scheier (1957)
89.0	174	11.31	Cairns and Scheier (1957)
82.0	46	2.78	Cairns and Scheier (1957)
82.0	174	11.23	Cairns and Scheier (1957)
90.0	200	15.20	INHS (unpublished)

9-1/8"

Percent saturation of dissolved oxygen was computed based upon the dissolved oxygen concentrations, altitude of the laboratory, and test temperatures. The hardness values of the Cairns and Scheier (1957) data were determined by calculation (Rund, Greenberg, and Taras 1976). The 96-hour LC50's of Cairns and Scheier (1957) were calculated by taking the mean of ranges.

The data presented above were used to develop a multiple regression equation which can be used to estimate the 96-hour LC50 at a given hardness and dissolved oxygen concentration. Here,

$$\text{96-hour LC50} = [0.182889\ (\text{D.O., \% saturation}) + 0.042461\ (\text{hardness, mg liter}^{-1}\ \text{as } CaCO_3 0)] - 12.96521$$

This equation is significant at the 0.05 level.

There are no data available regarding the effects of hydrogen ion concentration upon the tolerance of fish to zinc. However, hydrogen ion concentration governs the species and solubility of metals. At high pH many form hydroxides or basic carbonates which are relatively insoluble and tend to precipitate, but some of these compounds can remain in suspension. There

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is conflicting evidence regarding the relative toxicities of the various species of zinc. Suspended zinc was found to be nontoxic by Sprague (1964a, 1964b), equal to dissolved zinc in toxicity by Lloyd (1960), and more toxic than dissolved zinc by Mount (1966). The U.S. Environmental Protection Agency (1972), in light of such data, recommended that water quality criteria for metals be based upon total concentrations in water, regardless of chemical state or form.

Data from Cairns and Scheier (1957) indicate that temperature has no effect upon the 96-hour LC50 for zinc to bluegill.

The steps used for calculating the component toxicity due to zinc are as follows:

dissolved oxygen % saturation (DO%) = (DOPPM ÷ DO100) x 100

96-hour LC50, mg liter^{-1} as Zn, corrected for hardness and dissolved oxygen effects (CLC50) = [0.182889 (DO%) + 0.042461 (HARD)] - 12.96521

Note: if CLC50 $\leq$ 0, then CLC50 = 0.001

zinc component toxicity (ZNTOX) = ZN ÷ CLC50

where zinc concentration, mg liter^{-1} = ZN
hardness, mg liter^{-1} as $CaCO_3$ = HARD
dissolved oxygen, mg liter^{-1} = DOPPM
dissolved oxygen, mg liter^{-1} at 100% saturation = 0.97 x table value (Hutchinson, 1957, Table 74) = DO100

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SECTION 5

THE TEST WATERSHED

In order to develop the concept of the stress function it was necessary to actually apply, test, and varify the procedures described above on data obtained from a test watershed. The DuPage River basin in Northeastern Illinois was selected as the test watershed for the following reasons:

1. There is a wealth of biological, chemical, and physical data available for this stream;
2. The Northeastern Illinois Planning Commission has used this watershed in their 208 program and has a model of its hydrology and water quality active on the computer of the Circle Campus of the University of Illinois;
3. The basin is accessible readily to all of the parties involved in this research program;
4. The basin supports a wide range of water quality conditions from very poor to moderately good; and
5. The factors affecting water quality include agricultural non-point sources, municipal point sources, combined sewer overflows, and urban stormwater runoff.

The following pages describe in detail the DuPage River basin.

LOCATION AND BASIN MORPHOMETRY

The DuPage River basin is located in Cook, DuPage, and Will counties, Illinois (Fig. 6), and flows 117 km in a southerly direction, draining an area of approximately 91,427 ha via 462 km of permanent channels. The East Branch is a low-gradient stream (0.74m/km^{-1}). It essentially has become a stormwater drain and, especially, a carrier of wastewater treatment plant effluent for the surrounding suburban communities. There are approximately a dozen wastewater treatment plants and several oxidation lagoons which empty into the East Branch. The West Branch also is a low-gradient stream (0.96 m/km^{-1}). It primarily drains agricultural land plus extensive urbanized areas with numerous wastewater treatment plants. The West Branch flows through an intermorainal valley lying between the Minooka and Valparaiso moraines. The main branch of the DuPage River is formed by the confluence of the East and West Branches, approximately 2.5 km south of the DuPage-Will County line. It empties into the DesPlaines River near Channahon.

The DuPage River is an order 4 stream at its confluence with the DesPlaines River. Stream order is determined as follows: Unbranched

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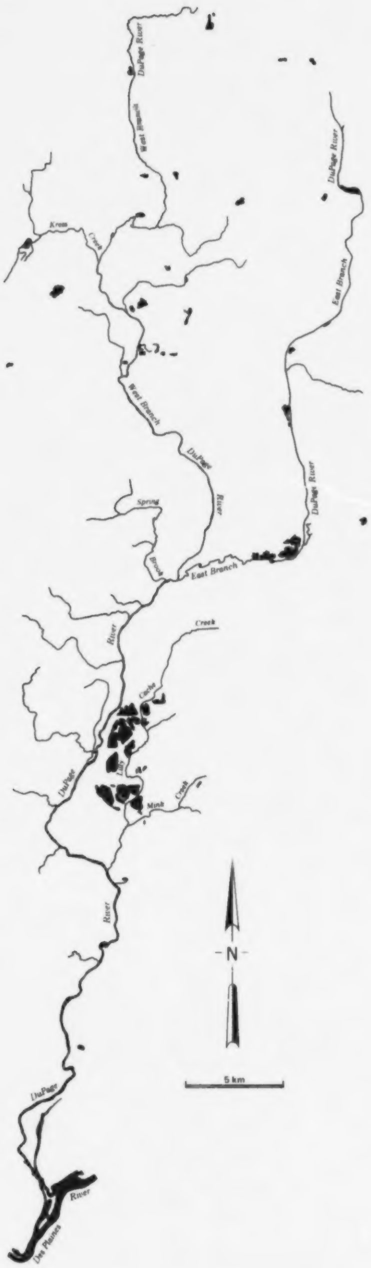
DuPage River
West Branch
Kress Creek
DuPage River
East Branch
West Branch
DuPage River
DuPage River
Spring Brook
East Branch
River
Creek
Cache
Lily
Mink Creek
DuPage
River
DuPage
River
Des Plaines
N
5 km

Figure 6. The test watershed, the DuPage River basin, Cook, DuPage, and Will Counties, Illinois.

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tributaries are order 1 streams; stream order is increased by one each time two streams of equal order join (i. e., two order 3 streams join to make an order 4 stream); stream order is unaffected when a lower order stream joins a higher order stream. One hundred and twenty-seven order 1 links have a mean length of 1.82 km and a total length of 230.98 km; 61 order 2 links gave a mean length of 1.86 km and a total length of 113.27 km; 45 order 3 links gave a mean length of 1.68 km and a total length of 75.67 km; and 19 order 4 links have a mean length of 2.19 km and a total length of 41.63 km. The drainage pattern of the DuPage River is intermediate between a fully bifurcate dendritic pattern and a trellis pattern. It is slightly underdeveloped at the order 1 and 2 levels and overdeveloped at the order 3 and 4 levels. It never reaches the potential order 8 stream possible if its drainage pattern was fully bifurcate dendritic.

The soils of the DuPage River watershed are developed from drift deposited late in the Wisconsin glaciation. Major soil associations include:

1) Drummer - Brenton - Proctor
2) Saybrook - Lisbon - Drummer
3) Zurich - Wauconda
4) Miami - Strawn
5) Lorenzo - Warsaw - Wea
6) Rodman - Casco - Fox
7) Sawmill - Lawson - Warsaw
8) Morley - Blount
9) Elliott - Ashkum - Varna
10) Toledo - Milford - Martinton
11) Elliott - Beecher - Markham
12) Bryce - Swygert - Frankfort

SUMMARY OF FISHERY DATA

The Metropolitan Sanitary District of Greater Chicago secured fishes from 15 sites along the DuPage River during 1976. The west and east branches of the river each were sampled at two locations. The remaining sites were located on the main branch of the river and its tributaries (Appendix A; Fig. 7). The river is known to have contained 54 species of fish and carp x goldfish hybrids (Appendix B). The 1976 Metropolitan Sanitary District of Greater Chicago collections included 37 species of fish and four hybrids (Appendix C). The hybrids were carp x goldfish, green sunfish x pumpkinseed, green sunfish x longear, and pumpkinseed x bluegill. Thirteen of the 17 fish species not included in the 1976 collections were taken only in collections made before 1905. The majority of the species remaining in the river are considered to have some tolerance to habitat abuse. Seven of the species currently found in the DuPage River are considered intolerant to habitat abuse. However, these intolerant fish represent only 4% of the total number of fish collected in the watershed. The most abundant species collected were green sunfish, bluntnose minnow, and spotfin shiner, which together represented 56% of the total number of fishes.

Throughout its length, the East Branch of the DuPage River contained few species of fish in small numbers. The headwaters of the West Branch also contained a poor quality fishery, but this reach of the river improved downstream. At the confluence of the two arms, the West Branch contained more species in numbers indicative of improved environmental conditions.

The tributaries of the DuPage River in which collections were taken illustrate their positive effect upon the river's fish population. Lilly Cache Creek and Hammel Creek sustain species expected in streams which have undergone little degradation. The similar diversity and equitability values for these areas are further proof of this condition. The low number of fishes collected from Station 136 and 137 may have resulted from inadequate collecting methods, since other collections in the immediate area, Hammel Creek for example, included many species. Fishery quality remained good throughout the river south of Lilly Cache Creek, with only a slight degrading influence from the Des Plaines River.

SUMMARY OF WATER QUALITY DATA

The DuPage River drainage system contained 30 Illinois Environmental Protection Agency water quality sites (Appendix D; Fig. 8). The Illinois Environmental Protection Agency had not analyzed for hardness at any of these sites in 1976. Also, eight sites were not analyzed for 16 or more parameters needed in the toxicity calculations. Two additional stations were not analyzed for MBAS. Toxicity index calculations indicated 10 stations having indices greater than 0.2 (Table 6). High ammonia nitrogen concentrations caused these values at the majority of the stations. Mercury and nickel were never detected in this system. Concentrations of arsenic, boron, cadmium, and chromium (trivalent and hecavalent) were present, yet they did not contribute to the stress function when rounded to the third decimal place. Those parameters which significantly contributed to stress were ammonia nitrogen, cyanide, and silver. Mean ammonia nitrogen values exceeded the Illinois

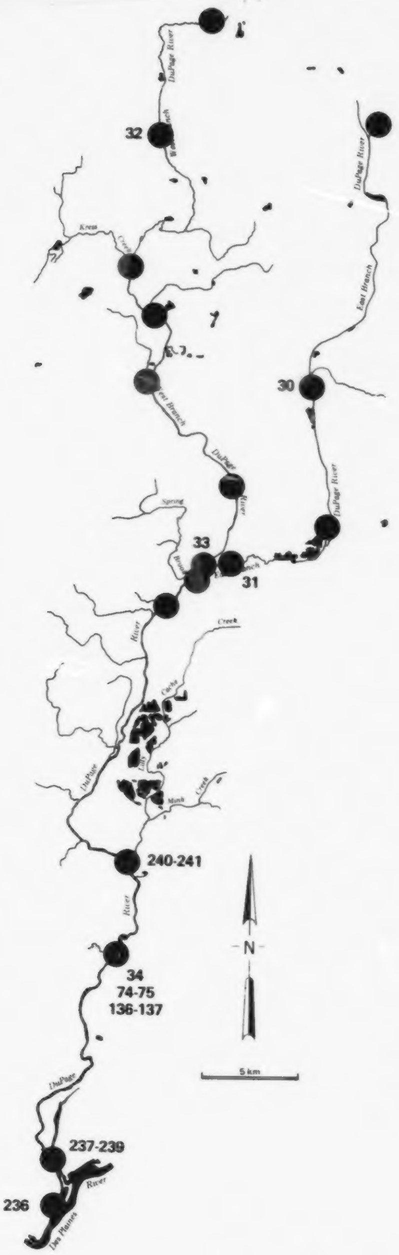
DuPage River
32
West Branch
DuPage River
Kress Creek
East Branch
30
East Branch
DuPage River
DuPage River
Spring
33
Branch
31
River
Creek
DuPage
Mink Creek
240-241
N
River
34
74-75
136-137
5 km
DuPage
237-239
River
236
Des Plaines

Figure 7. Location of fish sampling sites in the DuPage River basin, Cook, DuPage, and Will Counties, Illinois. Station numbers correspond to sampling sites of the 1976 Metropolitan Sanitary District of Greater Chicago fish inventory.

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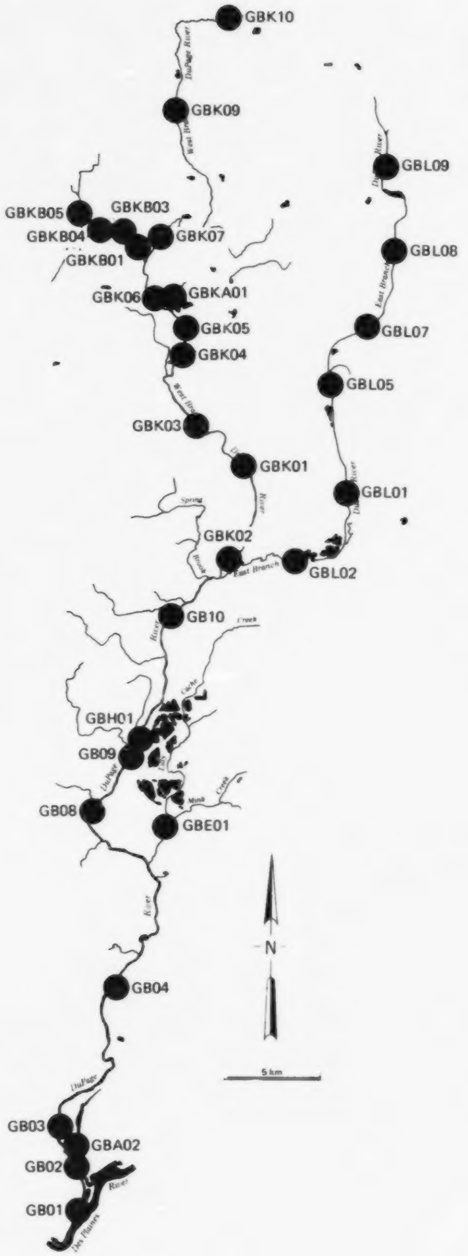
GBK10
DuPage River
GBK09
West Branch
GBL09
DuPage River
GBKB05
GBKB03
GBKB04
GBK07
GBKB01
GBL08
East Branch
GBK06
GBKA01
GBK05
GBL07
GBK04
GBL05
West Branch
GBK03
GBK01
River
GBL01
DuPage River
Spring
Brook
GBK02
East Branch
GBL02
GB10
River
Creek
GBH01
GB09
DuPage
Lily
Cache
Mink
Creek
GB08
GBE01
River
N
GB04
5 km
DuPage
GB03
GBA02
GB02
River
GB01
Des Plaines

Figure 8. Location of water quality sampling sites in the DuPage River basin, Cook, DuPage, and Will Counties, Illinois. Station numbers correspond to sampling sites of the Illinois Environmental Protection Agency.

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TABLE 6. BLUEGILL TOXICITY INDICES DERIVED FROM MEAN VALUES OF 1976 ILLINOIS ENVIRONMENTAL PROTECTION AGENCY DATA FROM SITES IN THE DUPAGE RIVER BASIN, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS

	Station	BGTU
DuPage River - West Branch	GBK10	.386
	GBK09	.111
	GBK07	.060
Kress Creek	GBKB05	.363
	GBKB04	.060
	GBKB03	.065
	GBLB01	.140
DuPage River - West Branch	GBK06	.098
Spring Brook	GBKA01	.448
DuPage River - West Branch	GBK05	.354
	GBK04	.124
	GBK03	.160
	GBK01	.069
	GBK02	.308
DuPage River - East Branch	GBL09	.554
	GBL08	.788
	GBL07	.678
	GBL05	2.204
	GBL01	.434
	GBL02	.134
DuPage River	GB10	.137
Norman Drain	GBH01	.008
DuPage River	GB09	.089
	GB08	.113
Lilly Cache Creek	GBE01	.060
DuPage River	GB04	.053
	GB03	.074
Illinois and Michigan Canal	GBA02	.058
DuPage River	GB02	.081
	GB01	.070

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Environmental Protection Agency standard at 15 stations (Appendix E). Cyanide never exceeded the Illinois Environmental Protection Agency standard, and silver exceeded the standard only at one station. At all the sites where copper was measured, the concentration equalled or exceeded the standard, yet it never contributed significantly to stress. Two other parameters, insignificant to the stress function, lead and fluoride, exceeded the Illinois Environmental Protection Agency standards at a few sites.

The East Branch of the DuPage River contained toxic concentrations of ammonia nitrogen throughout most of its length. Extreme conditions existed at the Lisle station (GBL05), where not only high ammonia nitrogen concentrations were encountered, but also significant concentrations of cyanide and silver. Two additional sites had high concentrations of MBAS. Water quality was not monitored extensively at the most downstream site of this arm. In spite of this, ammonia nitrogen concentrations were sufficiently high as to yield a high toxicity index indicative of high stress.

The West Branch of the DuPage River receives toxic ammonia nitrogen concentrations at its extreme headwaters. Water quality improves when Kress Creek joins with the West Branch. Kress Creek also receives an initial toxic ammonia nitrogen concentration, but rapidly recovers. The next tributary to enter the West Branch, Spring Brook, was not totally analyzed by the Illinois Environmental Protection Agency. Yet, those parameters that were measured indicated toxic ammonia nitrogen concentrations. The toxic conditions in this tributary dominated the river's quality through the next two downstream sampling sites. The second station downstream was not totally monitored, but significantly high ammonia nitrogen levels were recorded.

Water quality improved in the remaining stretch of the West Branch. Only one toxic input was recorded in this area. A toxic concentration of cyanide was detected in one sample from station GBK02.

The water quality of the main channel of the DuPage River resulted from the poor conditions present in the East Branch and the fair but improving conditions in the West Branch. The main channel continued to improve downstream primarily due to the many unaltered tributaries which flow into the river along the remainder of its length.

Lilly Cache Creek, one of these tributaries, originates in an urban area and flows south receiving water from several lakes and flooded gravel pits which greatly enhanced the creek's water quality upstream from its confluence with the river.

Favorable water quality conditions prevailed through the lower portion of the DuPage River. Environmental quality deteriorated slightly at the river's mouth due to the influence of the Illinois and Michigan canal and the Des Plaines River.

BIOLOGICAL COMMUNITIES

The presence or absence of a species at a site depends, in part, upon whether or not conditions there are suitable for its survival. The principal survival criterion is whether or not environmental variables fall within the species' range of tolerance. This relationship between organisms and their environment is a complex one, with each species having upper and lower lethal levels and an optimum level for every environmental variable.

For a given environmental variable, species tend to "sort out" along the continuum from the minimum to the maximum level for that variable. This ocurs because species generally have different ranges of tolerance and optima for any given environmental variable, and, in the case of overlapping ranges of tolerance, differences in optima frequently give one species the competitive edge over another. Thus, for water temperature, trout and some deep lake fishes are at their optimum in cold water and 10°C may represent an upper lethal limit for them. In contrast, species such as bass and many sunfish are at their optimum around 25°C and can survive even higher water temperatures. Still other species, such as carp, gar, and some topminnows, may survive water temperatures as high as 40°C.

The species associated with each habitat frequently are sufficient to call to mind a series of environmental variables and their appropriate ranges. Hence, the terms "trout stream" and "bass pond" convey physical and chemical information as well as biological information. In these examples, it is significant to note also that a more or less isolated biological observation is used to summarize a complex set of physical, chemical, and biological interactions. "Trout stream" recalls high water velocity, freedom from turbidity, low water temperature, and eroding substrate, low levels of nutrients and other dissolved solids, dissolved oxygen at or above saturation, few other fish species (dace, sculpins) and a benthic community dominated by mayflies, stoneflies, and caddisflies.

These same concepts have been applied in reviewing information on the distribution of fishes and similarities in water quality in the streams of northeastern Illinois (Brigham, McCormick, and Wetzel 1978). These streams sort, without too much difficulty, into five habitats as follows:

trout
pike
bass
carp
no fish

Table 7 summarizes all available information regarding the fishes of the DuPage River basin. The dates used in this table indicate the principal sources of data, as follows: pre-1908, Forbes and Richarson (1908); post-1950, Smith (1979); and 1976, the 1976 inventory of the Metropolitan Sanitary District of Greater Chicago. The 1976 data were not included in Smith's summary. The data in Table 7 are summarized under three headings, (bass, carp, and no fish), corresponding to communities now characteristic of that reach of the river (Fig. 9).

The summaries at the bottom of Table 7 indicate that the number of fish species in each reach of the river is declining. The summaries also indicate that each reach probably has maintained a comparable water quality relative to

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TABLE 7. FISHES KNOWN OR LIKELY TO OCCUR AT BASS, CARP, AND NO FISH SITES IN THE DUPAGE RIVER BASIN, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS, PRIOR TO 1908, SINCE 1950, AND IN 1976

Species	Bass			Carp			No Fish		
	1908	1950	1976	1908	1950	1976	1908	1950	1976
Grass pickerel	x	–	–	x	–	–	–	–	–
Northern pike	x	–	–	x	–	–	–	–	–
Stoneroller	x	x	x	x	x	–	x	–	–
Goldfish	–	–	–	–	x	–	–	–	–
Carp	–	x	x	–	x	x	–	–	–
Hornyhead chub	x	x	x	x	–	–	x	–	–
Golden shiner	–	x	x	–	–	–	–	–	–
Emerald shiner	x	x	x	x	–	–	–	–	–
Striped shiner	x	x	x	x	–	–	x	–	–
Bigmouth shiner	–	x	x	–	x	–	–	–	–
Blackchin shiner	x	–	–	x	–	–	–	–	–
Blacknose shiner	x	–	–	x	–	–	x	–	–
Spotfin shiner	–	x	x	–	x	–	–	–	–
Sand shiner	–	x	x	–	x	x	–	–	–
Redfin shiner	x	x	–	x	x	–	–	–	–
Bluntnose minnow	x	x	x	x	x	x	x	–	–
Fathead minnow	–	x	x	–	x	–	–	–	–
Creek chub	x	x	x	x	–	x	x	–	–
White sucker	x	x	x	–	–	x	–	–	–
Creek chubsucker	x	–	–	x	–	–	–	–	–
Northern hog sucker	x	–	–	x	–	–	–	–	–
Golden redhorse	x	–	–	x	–	–	–	–	–
Black bullhead	x	x	x	x	–	–	x	–	–
Yellow bullhead	x	–	–	x	–	–	x	–	–
Stonecat	–	x	–	–	x	–	–	–	–
Tadpole madtom	x	–	–	x	–	–	–	–	–
Blackstripe topminnow	x	–	–	x	–	–	x	–	–
Starhead topminnow	x	–	–	x	–	–	x	–	–
Brook silverside	x	–	–	x	–	–	–	–	–
Rock bass	x	x	–	x	x	–	–	–	–
Green sunfish	–	x	x	–	x	x	–	–	–
Pumpkinseed	–	x	–	–	x	–	–	–	–
Bluegill	–	x	x	–	x	–	–	–	–
Smallmouth bass	x	–	–	x	–	–	–	–	–
Black crappie	x	x	x	–	–	–	–	–	–
Mud darter	x	–	–	x	–	–	–	–	–
Rainbow darter	x	–	–	x	–	–	–	–	–
Least darter	x	–	–	x	–	–	–	–	–
Johnny darter	x	–	–	x	–	–	x	–	–
Banded darter	x	–	–	x	–	–	–	–	–
Slenderhead darter	x	–	–	x	–	–	–	–	–
Total species	31	21	17	28	14	6	12	0	0
Combined species		41			39			12	

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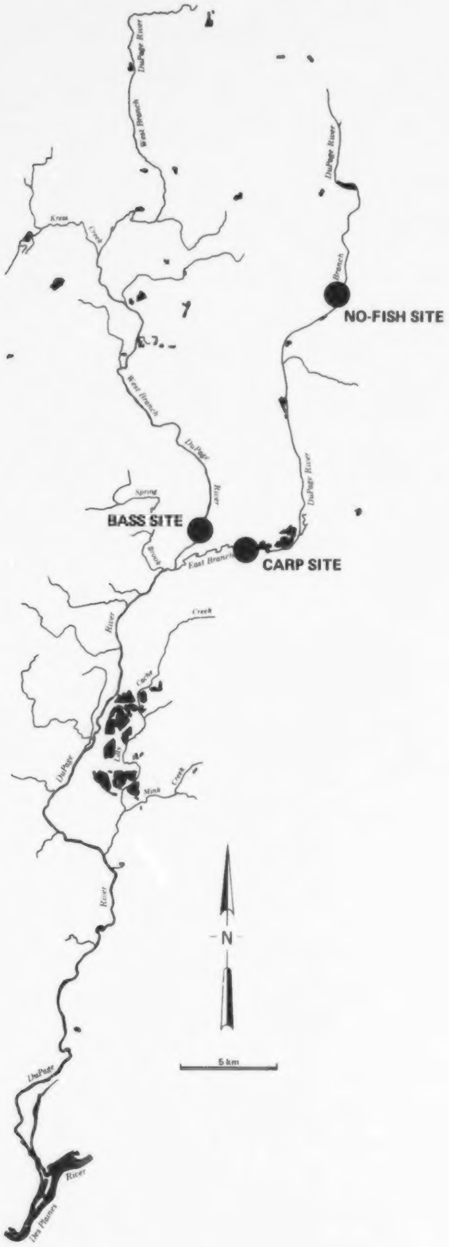
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Branch
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DuPage River
Spring
Brook
BASS SITE
East Branch
CARP SITE
River
Creek
Cache
Lily
DuPage
Mink Creek
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Des Plaines

Figure 9. Location of the Bass, Carp, and No Fish Sites in the DuPage River basin, Cook, DuPage; and Will Counties, Illinois.

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the rest of the basin through the period of record. Thus, the bass site on the West Branch in 1976 was roughly comparable to the carp site as it was in the 1950's and the no fish site as it was in the early 1900's. These data (Table 7) indicate that the West Branch and lower East Branch were pike waters in the early 1900's, but had degraded to a bass community by the middle of the century. Further habitat degradation in the lower East Branch in the past quarter century reduced this reach to carp habitat. The upper East Branch apparently was somewhat degraded at the turn of the century and even then did not support a pike community. Habitat degradation there was rapid and complete. The upper East Branch went from a bass community to no fish between 1908 and the 1950's. This reach remains degraded.

In order to perform simulations of water quality and to generate time series stress functions, it was necessary to select representative sites within each of the three existing habitat types in the DuPage River basin. These are located and described briefly below.

Bass Habitat

Representative Site: Illinois, Will County,
West Branch DuPage River 4 km SSE Naperville
upstream from Naperville Road bridge
T. 37N., R. 10 E., NE 1/4. section 6
Fish station 33, water quality station GBK02

Physical Description: Width - approximately 12 to 15 m
Depth - most littoral areas less than 0.5 m, main-flow area estimated 1 m
Substrate - silt-sand in littoral areas, cobble and rock in stronger currents
Banks - hard clay and mud, some rock, gradual drop-off
Riffle/Pool - flooded riffle in main flow, an isolated pool along northwest bank, a large expanse of swiftly flowing shallow pool throughout reach
Riparian - a heavily timbered southeast bank, many scattered trees along northwest bank, park land beyond.

Biological Description: Fish sampling at this site during 1976 yielded 223 fishes representing 17 species. Bluntnose minnows (89 individuals) and carp (40 individuals) were the most abundant species, representing more than half of the total number of fishes taken. Species present, in order of abundance, were:

Bluntnose minnow (89)
Carp (40)
Bigmouth shiner (21)
Green sunfish (20)
Spotfin shiner (16)
Sand shiner (7)
Creek chub (6)
Emerald shiner (4)
Fathead minnow (4)

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Bluegill (4)
Striped shiner (3)
White sucker (2)
Black bullhead (2)
Stoneroller (1)
Hornyhead chub (1)
Golden shiner (1)
Black crappie (1)
Sunfish hybrid (1)

Other species likely to be present at this site, but not taken in 1976, include redfin shiners, northern hogsuckers, stonecats, tadpole madtoms, blackstripe topminnows, smallmouth bass and largemouth bass. These species are common in similar habitats elsewhere in northeastern Illinois and many are known from this reach of the DuPage River from earlier sampling.

Carp Habitat

Representative Site: Illinois, Will County,
East Branch DuPage River 5 km SSE Naperville
upstream from Naperville Road bridge
T. 37 N., R. 10 E., NE 1/4, section 7
Fish site 31, near water quality site GBL02

Physical Description: Width - approximately 4.5 m
Depth - estimated 1 m, uniform
Substrate - hard clay and mud
Banks - hard clay and mud
Riffle/Pool - none, channel dredged (perhaps within 20 to 25 years)
Riparian - a narrow belt of timber along both banks, row-crop agriculture beyond.

Biological Description: Fish sampling at this site during 1976 yielded 25 fishes representing 6 species. Carp (8 individuals) and green sunfish (7 individuals) were the most abundant species, representing more than half of the total number of fishes taken. Species present, in order of abundance, were:

Carp (8)
Green sunfish (7)
Bluntnose minnow (5)
Sand shiner (2)
Creek chub (2)
White sucker (1)

Other species likely to be present at this site, but not taken in 1976, include goldfish and black bullheads. These species are common in similar habitats elsewhere in northeastern Illinois.

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No-Fish Habitat

Representative Site: Illinois, DuPage County,
East Branch DuPage River in east Glen Ellyn
0.3 km W Illinois highway 53
T. 39 N., R. 10 E., center, section 13
Near fish site 30, water quality site GBL05

Physical Description: Width - approximately 3 m
Depth - approximately 1 m, uniform
Substrate - hard clay and mud
Banks - hard clay and mud, straight drop-off
Riffle/Pool - none, channel recently dredged (perhaps within 5 to 10 years)
Riparian - a narrow belt of timber and suburban habitat to the west; old field habitat to the east

Biological Description: Fishes are not known to inhabit this reach of the East Branch of the DuPage River. Sampling during 1976 and by seining on 5 December 1978 did not yield fishes at this site. Oligochaeta and Chironomidae were found to be abundant in 1978.

In spite of the absence of fishes, the physical habitat does not appear to be limiting to the survival of tolerant species. Cover and food organisms are available in moderate amounts. Suitable spawning sites, however, were not present.

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SECTION 6

APPLICATIONS

UNALTERED RUN

Computer simulation of the stress function was performed for a 3-year period (October 1970 to September 1973) for the three study sites described above. The following parameters were included as component toxicities in the calculation of the total stress at each site because it was known (Appendix E) that they were present in significant concentrations in at least one site:

Ammonia

Cyanide

Lead

Zinc

Copper

LAS

Residual Chlorine

A summary of the stress function is presented in Table 8, where the maximum, minimum, and mean stress for each site is given. Also listed are the component toxicities, which by definition must collectively equal the total stress.

Inspection of the mean total stress at each site reveals that the levels of stress the fish are subjected to are quite high at the no fish site, significantly lower at the bass site, and intermediate at the carp site. These results are in line with those that had been initially predicted. The no fish site is unable to maintain any fish population because the stress placed upon a population there exceeds the tolerance limit for survival. The bass site, however, is able to maintain a diverse fishery because the stress levels there are comparatively much less. A carp community is able to withstand a limited, but significant degree of water quality degradation.

The extremely high mean stress value at the no fish site is due to relatively high levels of ammonia, LAS, and residual chlorine, in combination with very low dissolved oxygen concentrations, particularly during summer

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TABLE 8. MAXIMUM, MINIMUM, AND MEAN VALUES FOR THE STRESS FUNCTION AND COMPONENT TOXICITIES AT THREE SITES IN THE DUPAGE RIVER BASIN, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS, WITH UNALTERED DATA AND WITH MODIFICATIONS (SEE TEXT) IN DISSOLVED OXYGEN, HARDNESS, WATER TEMPERATURE, AND pH

		Modified Parameters			
	Unaltered Run	Diss. Oxy.	Hard.	Water Temp.	pH
No Fish Site					
Total Stress					
max.	783.7	783.7	783.7	766.9	1034.5
min.	0.120	0.109	0.120	0.116	0.125
mean.	23.024	6.976	23.032	22.514	50.364
Component Toxicity					
Ammonia	20.18	4.759	20.18	19.56	47.52
Cyanide	0.088	0.082	0.088	0.087	0.088
Lead	0.023	0.020	0.023	0.023	0.023
Zinc	0.004	0.003	0.011	0.005	0.004
Copper	0.020	0.018	0.021	0.020	0.020
LAS	0.772	0.157	0.772	0.918	0.772
Res. chlorine	1.937	1.937	1.937	1.901	1.937
Carp Site					
Total Stress					
max.	98.47	114.6	149.6	96.20	135.5
min.	0.155	0.169	0.157	0.154	0.166
mean.	1.039	1.197	1.046	1.035	1.195
Component Toxicity					
Ammonia	0.116	0.211	0.116	0.117	0.272
Cyanide	0.081	0.085	0.081	0.079	0.081
Lead	0.016	0.018	0.016	0.018	0.016
Zinc	0.046	0.050	0.052	0.046	0.046
Copper	0.020	0.022	0.021	0.021	0.020
LAS	0.107	0.158	0.107	0.109	0.107
Res. chlorine	0.653	0.653	0.653	0.645	0.653
Bass Site					
Total Stress					
max.	0.279	26.28	24.84	0.279	0.332
min.	0.005	0.006	0.005	0.005	0.006
mean.	0.115	0.137	0.120	0.114	0.136
Component Toxicity					
Ammonia	0.014	0.020	0.014	0.014	0.035
Cyanide	0.000	0.000	0.000	0.000	0.000
Lead	0.000	0.000	0.000	0.000	0.000
Zinc	0.003	0.011	0.007	0.003	0.003
Copper	0.008	0.008	0.009	0.008	0.008
LAS	0.030	0.038	0.030	0.030	0.030
Res. chlorine	0.060	0.060	0.060	0.059	0.060

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months. It was observed that as the percent saturation of dissolved oxygen was reduced, the component toxicity for ammonia (as well as for zinc, lead, copper, and LAS) became particularly sensitive. For example, when the saturation level is equal to or less than 24.78%, the CLC50 in the ammonia toxicity equation reaches zero, and the computer program defaults the CLC50 value to 0.001. At this point, the ammonia toxicity rises by several orders of magnitude. The maximum stress level at the site, 783.7, occurred at a particular hour when dissolved oxygen reached zero. Municipal wastewater treatment plants located upstream of the no fish site are the primary cause of the significant levels of ammonia toxicity (mean 20.18), residual chlorine toxicity (mean 1.937), and LAS toxicity (mean 0.7729).

The bass site typically had much higher dissolved oxygen and lower ammonia concentrations in the summer months, so that its ammonia component was much lower than that of the no fish site. There was also less treatment plant contribution at the bass site, so that the residual chlorine and LAS toxicities were lower. Lead toxicity at this site was 0.000, probably a result of the higher dissolved oxygen concentration (lead toxicity is a function of dissolved oxygen).

The carp site was located a significant distance downstream from most of the larger East Branch treatment plants and this was reflected in its stress function. The mean ammonia stress contribution, due to a combination of lower ammonia levels and improved dissolved oxygen concentrations, was substantially lower than at the no fish site upstream. The residual chlorine and LAS toxicities also were substantially lower than those at the no fish site, due to the effects of dilution and/or degradation. Though dissolved oxygen concentrations were less critical than at the no fish site, they did dip low enough on occasion to cause extremely high levels of ammonia toxicity. Another observation was that the mean zinc toxicity (0.046) was really not representative of the typical daily mean, which over the 3-year period was about 0.004. The mean for the 3-year period was high because of a 0.0 mg liter^{-1} dissolved oxygen concentration in August 1972 which lasted 20 hours. During that event, the zinc toxicity averaged about 30, thereby contributing strongly to the high 3-year mean.

As stated earlier in this report, other characteristics of the stress function may be described in terms of the frequency and duration of its excursions about an arbitrary value called the stress index. Tables 9 through 11 characterize the stress functions at the no fish, carp, and bass sites for excursions of 1-, 24-, and 96-hours duration relative to stress indices ranging from 0.2 to 4.0. Again, these data are based upon hourly values for the stress function for a 3-year period. The table data are read as follows:

> The stress function at the bass site showed two excursions above a stress index of 0.250 that had durations of 24 hours or more. The average length of these excursions was 36.5 hours. In contrast, the stress function at this site showed six excursions below the stress index of 0.250 that had durations of 24 hours or more. The average length

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TABLE 9. FREQUENCY AND MEAN DURATION (HOURS IN PARENTHESES) OF POSITIVE AND NEGATIVE EXCURSIONS OF THE STRESS FUNCTION ABOUT STRESS INDICES FROM 0.2 THROUGH 4.0 DURING THE PERIOD OCTOBER 1970 THROUGH SEPTEMBER 1973 AT THE NO FISH SITE, DUPAGE RIVER, DUPAGE COUNTY, ILLINOIS

Stress Index	Positive Excursions			Negative Excursions		
	1	24	96	1	24	96
0.2	4 (6572)	4 (6572)	4 (6572)	3 (5)	0 (0)	0 (0)
0.4	12 (2186)	11 (2384)	10 (2613)	11 (7)	0 (0)	0 (0)
0.6	28 (929)	25 (1039)	23 (1126)	27 (11)	5 (35)	0 (0)
0.8	58 (440)	42 (606)	33 (754)	57 (14)	9 (51)	0 (0)
1.0	73 (339)	51 (480)	42 (570)	72 (22)	20 (59)	3 (113)
1.2	94 (251)	62 (376)	38 (579)	94 (29)	35 (63)	6 (119)
1.4	103 (216)	67 (328)	44 (468)	103 (39)	38 (91)	12 (160)
1.6	134 (155)	68 (298)	44 (431)	135 (41)	39 (117)	21 (173)
1.8	147 (130)	64 (286)	38 (447)	148 (49)	42 (144)	22 (230)
2.0	155 (111)	55 (297)	34 (449)	156 (58)	42 (187)	22 (308)
2.2	147 (107)	59 (253)	33 (407)	148 (72)	42 (226)	19 (437)
2.4	173 (83)	50 (265)	31 (395)	174 (69)	38 (278)	18 (517)
2.6	177 (74)	47 (354)	32 (349)	178 (75)	40 (293)	20 (523)
2.8	191 (61)	47 (222)	22 (416)	192 (76)	36 (360)	21 (575)
3.0	194 (55)	39 (232)	21 (386)	195 (81)	39 (361)	17 (756)
3.2	212 (45)	39 (197)	18 (360)	213 (79)	40 (375)	20 (697)
3.4	216 (39)	37 (180)	15 (358)	217 (82)	38 (419)	18 (823)
3.6	242 (31)	36 (147)	12 (331)	243 (78)	44 (389)	17 (929)
3.8	243 (27)	32 (134)	9 (340)	244 (81)	39 (459)	18 (929)
4.0	241 (25)	31 (113)	10 (246)	242 (84)	43 (434)	18 (959)

TABLE 10. FREQUENCY AND MEAN DURATION (HOURS IN PARENTHESES) OF POSITIVE AND NEGATIVE EXCURSIONS OF THE STRESS FUNCTION ABOUT STRESS INDICES FROM 0.2 THROUGH 4.0 DURING THE PERIOD OCTOBER 1970 THROUGH SEPTEMBER 1973 AT THE CARP SITE, DUPAGE RIVER, WILL COUNTY, ILLINOIS

Stress Index	Positive Excursions			Negative Excursions		
	1	24	96	1	24	96
0.2	3 (8754)	3 (8754)	3 (8754)	2 (21)	1 (24)	0 (0)
0.4	23 (1120)	20 (1288)	18 (1426)	22 (24)	8 (50)	0 (0)
0.6	112 (196)	53 (403)	33 (616)	113 (38)	42 (88)	14 (160)
0.8	150 (72)	36 (275)	22 (415)	151 (101)	67 (214)	36 (352)
1.0	74 (89)	31 (201)	14 (386)	75 (262)	37 (521)	26 (719)
1.2	57 (73)	21 (181)	11 (294)	58 (381)	31 (705)	20 (1061)
1.4	41 (62)	19 (125)	8 (228)	42 (565)	29 (814)	23 (1016)
1.6	42 (32)	11 (98)	4 (157)	43 (580)	27 (918)	21 (1166)
1.8	25 (21)	5 (63)	1 (157)	26 (991)	19 (1353)	16 (1597)
2.0	16 (10)	1 (44)	0 (0)	17 (1538)	13 (2007)	12 (2171)
2.2	11 (11)	1 (44)	0 (0)	12 (2182)	11 (2380)	10 (2615)
2.4	11 (11)	1 (44)	0 (0)	12 (2182)	11 (2380)	10 (2615)
2.6	11 (11)	1 (44)	0 (0)	12 (2182)	11 (2380)	10 (2615)
2.8	12 (10)	1 (43)	0 (0)	13 (2014)	11 (2381)	10 (2615)
3.0	12 (10)	1 (42)	0 (0)	13 (2015)	11 (2381)	10 (2615)
3.2	11 (10)	1 (42)	0 (0)	12 (2183)	10 (2619)	9 (2906)
3.4	11 (10)	1 (42)	0 (0)	12 (2183)	10 (2619)	9 (2906)
3.6	11 (10)	1 (42)	0 (0)	12 (2183)	10 (2619)	9 (2906)
3.8	11 (10)	1 (42)	0 (0)	12 (2183)	10 (2619)	9 (2906)
4.0	11 (10)	1 (42)	0 (0)	12 (2183)	10 (2619)	9 (2906)

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TABLE 11. FREQUENCY AND MEAN DURATION (HOURS IN PARENTHESES) OF POSITIVE AND NEGATIVE EXCURSIONS OF THE STRESS FUNCTION ABOUT STRESS INDICES FROM 0.05 THROUGH 0.50 DURING THE PERIOD OCTOBER 1970 THROUGH SEPTEMBER 1973 AT THE BASS SITE, DUPAGE RIVER, WILL COUNTY, ILLINOIS

Stress Index	Positive Excursions			Negative Excursions		
	1	24	96	1	24	96
0.05	1 (354)	26 (962)	17 (1434)	71 (12)	5 (55)	1 (127)
0.10	72 (66)	29 (401)	15 (729)	195 (69)	63 (196)	24 (423)
0.15	194 (113)	17 (359)	12 (479)	57 (351)	30 (661)	17 (130)
0.20	56 (91)	25 (131)	11 (231)	38 (604)	24 (949)	12 (1843)
0.25	37 (15)	2 (37)	0 (0)	10 (2617)	6 (4357)	6 (4357)
0.30	9 (0)	0 (0)	0 (0)	1 (6304)	1 (6304)	1 (6304)
0.35	0 (0)	0 (0)	0 (0)	1 (6304)	1 (6304)	1 (6304)
0.40	0 (0)	0 (0)	0 (0)	1 (6304)	1 (6304)	1 (6304)
0.40	0 (0)	0 (0)	0 (0)	1 (6304)	1 (6304)	1 (6304)
0.50	0 (0)	0 (0)	0 (0)	1 (6304)	1 (6304)	1 (6304)

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of these excursions was 4356.8 hours.

After examination of the basic stress function results, several tests of the function's sensitivity to the levels or concentrations of several input constituents were performed. The sensitivity runs involved varying the concentration or level of a constituent as it entered into the stress function (e. g. temperature). Note that no change was made in the original QUALITY simulation. For example, a change in temperature would, in reality, impact upon the concentrations of other constituents, such as dissolved oxygen and ammonia, in QUALITY. However, in the context of this sensitivity study, only the direct effect of temperature upon the toxicity formulation is considered.

The first of the modifications was to dissolved oxygen concentration. Because very low dissolved oxygen levels were found to be quite critical in the calculation of the component toxicities at the no fish site, the dissolved oxygen concentrations input to the stress function were increased here by 50%. On the other hand, dissolved oxygen levels were much higher at the carp and bass sites, so the sensitivity of the function to a modest decrease (20%) in DO levels was investigated. Table 8 summarizes these results. The no fish station, where dissolved oxygen was increased, exhibited a substantial decrease in overall mean stress, primarily due to the decrease in the ammonia and LAS component toxicities. At the other two sites, the 20% decrease in dissolved oxygen resulted in only a proportional increased in overall mean stress. The low sensitivity of the stress function at these sites to a modest decrease in dissolved oxygen seems to indicate a reasonable degree of stability in these ecosystems. An important consideration, however, is that the maximum stress here increased from 0.279 to 26.3, implying a potentially drastic impact upon fish survival.

The sensitivity of the stress function to changes in hardness and water temperature also were analyzed. Again, note that for water temperature in particular, the changes in reaction rates of the basic quality run were not taken into account in these tests. Referring to Table 8, it is seen that neither a decrease of 10% in hardness, nor a decrease of 5% in water temperature have a significant impact upon overall mean stress. Again, though, the reduction in hardness had a substantial impact upon the maximum stress, due to the sensitivity of several component stress equations at low hardness levels (This situation is similar to that of dissolved oxygen discussed previously).

The last sensitivity test involved the increase of pH levels by 5%. Table 8 shows that the stress function at the no fish site was particularily sensitive to this degree of change. This was due to a substantial rise in the un-ionized ammonia concentration, resulting in a corresponding increase in stress due to ammonia. The carp and bass sites did not exhibit this degree of sensitivity in total stress, even though the relative change in the ammonia component was similar to the change at the no fish site.

A constituent of uncertain importance to fish toxicity in the DuPage River is silver. One Hundred and Eighty Five Illinois Environmental Protection Agency in-stream samples from the DuPage River between 1972 and 1977 were analyzed for the presence of silver. Of those 185 samples, only 6 contained silver concentrations greater than 0.000 mg liter^{-1}; the remaining 179 samples had reported concentrations of 0.000. These 6 values consisted of 4 values of 0.010 mg liter^{-1}, and 2 values of 0.020 mg liter^{-1}. When these values are used in the silver toxicity equation, the following silver component toxicities result:

CONCENTRATION	STRESS
.010 mg/l	2.56
.020 mg/l	5.12

At the concentrations listed above the silver component would certainly be very important in determining overall stress. However, because of the paucity of the data and the erratic concentrations reported, the silver component was not used in the stress function. The magnitude of the computed silver toxicity suggests the importance of including the silver component when reliable data are available. It also suggests the importance of accurately measuring silver concentrations below .01 mg liter^{-1}, which is apparently the current detection limit for the Illinois Environmental Protection Agency.

Several questions were raised in the simulation of the stress function. Perhaps the most significant of these is the severe effect of low dissolved oxygen concentrations. As pointed out earlier, the most influential component at the no fish site was ammonia during periods of low dissolved oxygen saturation. Table 12 illustrates one such low dissolved oxygen event. Note the rapid increse in stress around hour 16, when dissolved oxygen drops below the threshold of 24.78% saturation and the CLC50 in the toxicity equation becomes negative (and defaults to 0.001). The resultant high levels of stress (>100) would theoretically be quickly lethal to most fish.

Stress values of this magnitude appear to be an excessive indicator of stress to the fish, since such values should be lethal almost instantaneously. Several alternatives are possible in the stress program to eliminate these unwieldy values, such as placing a reasonable "ceiling" on the maximum value stress can attain. Another possibility would be to increase the CLC50 default value (0.001) to a somewhat larger number. However, any such modification might tend to mask the apparently severe effect low dissolved oxygen has upon the stress function. Since there are no experimental data to date to support a modification of the original calculations, the function will be left unchanged in respect to this issue.

Another issue raised in the stress analysis was that of high residual chlorine toxicity during winter months. Residual chlorine degradation occurs at a slower rate at cold temperatures than it does at warm temperatures. This results in typically higher simulated in-stream residual chlorine concentrations in winter than in summer. The toxicity of chlorine, however, appears to be substantially greater at high temperatures than at low temperatures, at least in the 20°C to 30°C range. For example, chlorine toxicity is over twenty times greater at 30°C than at 20°C for a given concentration of chlorine. This is the assumption used in the current stress function. However, the function also assumes that residual chlorine toxicity

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TABLE 12. SENSITIVITY OF THE STRESS FUNCTION TO LOW CONCENTRATIONS OF DISSOLVED OXYGEN AT THE NO FISH SITE, DUPAGE RIVER, DUPAGE COUNTY, ILLINOIS, ON 20 MAY 1972 (DISSOLVED OXYGEN REACHES THE THRESHOLD VALUE OF 24.78% SATURATION AT HOUR 16)

Hour	Stress Func.	Ammonia Toxicity	Dissolved Oxygen Conc.	Ammonia Conc.	Water Temp.	pH
1	3.799	2.038	2.4	0.111	18.4	7.8
2	3.557	1.764	2.5	0.098	18.2	7.8
3	3.461	1.621	2.5	0.094	17.9	7.8
4	2.946	1.068	2.6	0.091	17.7	7.8
5	2.809	0.883	2.7	0.087	17.5	7.8
6	2.620	0.660	2.8	0.087	17.5	7.8
7	2.585	0.589	2.8	0.089	17.8	7.8
8	2.606	0.577	2.8	0.092	18.2	7.8
9	2.658	0.619	2.8	0.098	18.6	7.8
10	2.731	0.720	2.8	0.108	19.0	7.8
11	2.826	0.855	2.7	0.117	19.4	7.8
12	2.935	1.015	2.6	0.125	19.8	7.8
13	3.350	1.482	2.5	0.133	20.2	7.8
14	3.983	2.121	2.4	0.138	20.5	7.8
15	4.970	3.099	2.3	0.138	20.9	7.8
16	140.094	138.188	2.1	0.138	21.2	7.8
17	136.054	134.112	2.0	0.134	21.2	7.8
18	129.780	127.806	1.9	0.128	21.2	7.8
19	124.018	122.036	1.9	0.122	20.9	7.8
20	121.532	119.574	1.9	0.120	20.5	7.8
21	121.170	119.254	1.9	0.119	20.1	7.8
22	120.186	118.270	2.0	0.118	19.2	7.8
23	116.987	115.074	2.0	0.115	19.4	7.8
24	112.727	110.807	2.1	0.111	19.4	7.8

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is constant below 20°C for a given chlorine concentration. This appears to be inconsistent with the radical toxicity difference between 20°C and 30°C. However, data on residual chlorine toxicity in the 0°C to 20°C range are quite limited and sometimes contradictory. In conclusion, the model's prediction of higher chlorine component stress in winter than in summer is questionable.

ALTERED RUN

The development of the stress function has made possible a means of directly relating fish diversity and survival with the chemical and physical components of an ecosystem. The use of the function as a predictive tool could enable one to determine the biological potential of a stream, as well as suggest pollution control strategies for realization of that potential. A hypothetical management plan to reduce the ammonia component at the no fish site was developed and incorporated into the stress function. The plan involved several strategies: 1) limit wastewater treatment plant effluent ammonia concentrations to 1.5 mg liter^{-1} during summer and 4.0 mg liter^{-1} during winter, 2) eliminate combined sewer overflows, 3) reduce sediment oxygen demand substantially, and 4) moderately increase dissolved oxygen concentrations in wastewater treatment plant effluents. Table 13 summarizes the results of this management test. The overall mean stress was reduced by more than order of magnitude, down to 2.12. Ammonia toxicity, which was the principal target of the management strategy, was reduced by nearly three orders of magnitude. The mean stress of 2.12, however, was still significantly higher than that of the carp site.

The results suggest that if a more diverse fishery than carp is desired, additional control strategies might need to be implemented. Upon inspection of the component toxicities, residual chlorine is now found to be the principal component of stress (1.935). If chlorine from wastewater treatment plant effluents could be reduced, the overall mean stress might be lowered to a level comparable to that found at the bass site. Presently, chlorination of wastewater treatment plant effluents is considered an important "finishing" step in the treatment process.

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TABLE 13. IMPACT OF A HYPOTHETICAL WATER QUALITY MANAGEMENT PLAN (SEE TEXT FOR EXPLANATION) UPON THE STRESS FUNCTION AT THE NO FISH SITE, DUPAGE RIVER, DUPAGE COUNTY, ILLINOIS

	Unaltered Run	Altered Run
Total Stress		
max.	783.7	51.48
min.	0.120	0.118
mean.	23.02	2.121
Component Toxicity		
Ammonia	20.18	0.023
Cyanide	0.088	0.079
Lead	0.023	0.000
Zinc	0.004	0.003
Copper	0.020	0.008
LAS	0.772	0.073
Res. chlorine	1.937	1.935

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APPENDIX A

LOCATION OF FISH SAMPLING SITES FOR THE 1976 METROPOLITAN SANITARY DISTRICT OF GREATER CHICAGO INVENTORY ON THE DUPAGE RIVER, COOK, DUPACE, AND WILL COUNTIES, ILLINOIS

Number	Stream	Location
30	East Branch, DuPage River	T38N/R 10E/S 15 NE; DuPage Co., immediately upstream and downstream of Maple Ave., Lisle.
31	East Branch, DuPage River	T37N/R 10E/S 7 NE; Will Co.; upstream of Naperville Road, Naperville.
32	West Branch, DuPage River	T40N/R 9E/S 35 NW; DuPage Co,; immediately upstream and downstream of Rt 64 (North Ave.), West Chicago.
33	West Branch, DuPage River	T37N/R 10E/S; 6 NE; Will Co.; 200 meters upstream of Naperville Rd, Naperville.
34	DuPage River	T35N/R 9E/S 10SE; Will Co., 40 meters and 60 meters above Hammel Woods Dam, Shorewood.
74	Hammel Creek	T35N/R 9E/S 10 S/E; Will Co.; mouth of Hammel Creek with DuPage River, Hammel Creek, Shorewood.
75	Hammel Creek	T35N/R 9E/S 10 SE; Will Co.; 100 meters above with DuPage River, Hammel Woods, Shorewood.
136	DuPage River	T35N/R 9E/S 10 SE; Will Co.; forebay and tailrace of stone dam, Shorewood Park off Rt 52, Shorewood.
137	DuPage River	T35N/R 9E/S 10 SE; Will Co.; 100 meters below stone dam, Shorewood Park off Rt 52, Shorewood.
236	DuPage River	T34N/R 9E/S 20 NE; Will Co.; mouth of DuPage River with Des Plaines River, Channahon.
237	DuPage River	T34N/R 9E/S 17 SW; Will Co.; 150 meters downstream of Channahon St Pk Dam, Channahon.

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238	DuPage River	T34N/R 9E/S 17 SW; Will Co.; tailrace and forebay of Channahon St Pk Dam, Channahon.
239	DuPage River	T34N/R 9E/S 17 SW; Will Co.; 120 meters below Channahon St Pk Dam, Channahon.
240	Lilly Cache Creek	T36N/R 9E/S 27 SE; Will Co.; mouth of Lilly Cache Creek junc. DuPage River, Plainfield.
241	Lilly Cache Creek	T36N/R 9E/S 27 SE; Will Co.; 100 meters above junc. with DuPage River, Plainfield.

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APPENDIX B

CHECKLIST OF FISHES KNOWN TO OCCUR IN THE DUPAGE RIVER, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS.

Clupeiformes

Clupeidae - Herrings

Dorosoma cepedianum (Lesueur)
Gizzard shad

Salmoniformes

Umbridae - Mudminnows

Umbra limi (Kirtland)
Central mudminnow

Esocidae - Pikes

Esox americanus vermiculatus Lesueur
Grass pickerel

Esox lucius Linnaeus
Northern pike

Cypriniformes

Cyprinidae - Minnows and Carps

Campostoma anomalum (Rafinesque)
Stoneroller

Campostoma oligolepis Hubbs & Green
Largescale stoneroller

Carassius auratus (Linnaeus)
Goldfish

Cyprinus carpio (Linnaeus)
Carp

Cypriniformes (continued)

Cyprinidae - Minnows and Carps (continued)

Nocomis biguttatus (Kirtland)
Hornyhead chub

Notemigonus crysoleucas (Mitchill)
Golden shiner

Notropis atherinoides (Rafinesque)
Emerald shiner

Notropis chrysocephalus (Rafinesque)
Striped shiner

Notropis cornutus (Mitchill)
Common shiner

Notropis dorsalis (Agassiz)
Bigmouth shiner

Notropis heterolepis Eigenmann & Eigenmann
Blacknose shiner

Notropis lutrensis (Baird & Girard)
Red shiner

Notropis rubellus (Agassiz)
Rosyface shiner

Notropis spilopterus (Cope)
Spotfin shiner

Notropis stramineus (Cope)
Sand shiner

Notropis umbratilis (Girard)
Redfin shiner

Phenacobius mirabilis (Girard)
Suckermouth minnow

Phoxinus erythrogaster (Rafinesque)
Southern redbelly dace

Pimephales notatus (Rafinesque)
Bluntnose minnow

Pimephales promelas Rafinesque
Fathead minnow

Cypriniformes (continued)

Cyprinidae - Minnows and Carps (continued)

Pimephales vigilax (Baird & Girard)
Bullhead minnow

Semotilus atromaculatus (Mitchill)
Creek chub

Carp X Goldfish hybrid

Catostomidae - Suckers

Catostomus commersoni (Lacepede)
White sucker

Erimyzon oblongus (Mitchill)
Creek chubsucker

Hypentelium nigricans (Lesueur)
Northern hog sucker

Moxostoma duquesnei (Lesueur)
Black redhorse

Moxostoma erythrurum (Rafinesque)
Golden redhorse

Siluriformes

Ictaluridae - Freshwater catfishes

Ictalurus melas (Rafinesque)
Black bullhead

Ictalurus natalis (Lesueur)
Yellow bullhead

Noturus exilis Nelson
Slender madtom

Noturus flavus Rafinesque
Stonecat

Noturus gyrinus (Mitchill)
Tadpole madtom

Atheriniformes

Cyprinodontidae - Killifishes

Atheriniformes (continued)

Cyprinodontidae - Killifishes (continued)

Fundulus notatus (Rafinesque)
Blackstripe topminnow

Fundulus notti (Agassiz)
Starhead topminnow

Labidesthes sicculus (Cope)
Brook silversides

Perciformes

Centrarchidae - Sunfishes

Amboplites rupestris (Rafinesque)
Rock bass

Lempomis gibbosus (Linnaeus)
Pumpkinseed

Lepomis macrochirus Rafinesque
Bluegill

Lepomis megalotis (Rafinesque)
Longear sunfish

Micropterus dolomieui Lacepede
Smallmouth bass

Micropterus salmoides (Lacepede)
Largemouth bass

Pomoxis nigromaculatus (Lesueur)
Black crappie

Percidae - Perches

Etheostoma asprigene (Forbes)
Mud darter

Etheostoma flabellare Rafinesque
Fantail darter

Etheostoma microperca Jordan & Gilbert
Least darter

Etheostoma nigrum Rafinesque
Johnny darter

Perciformes (continued)

Percidae - Perches (continued)

Etheostoma zonale (Cope)
Banded darter

Percina maculata (Girard)
Blackside darter

APPENDIX C

ABUNDANCE OF FISHES AT SAMPLING SITES IN THE DUPAGE RIVER, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS, DURING THE 1976 METROPOLITAN SANITARY DISTRICT OF GREATER CHICAGO INVENTORY. SPECIES DIVERSITY (d) AND EQUITABILITY (e) ARE EXPRESSED FOR EACH STATION

	STATIONS															TOTAL NUMBER	% OF TOTAL
	30	31	32	33	241	240	34	74	75	136	137	237	238	239	236		
CLUPEIDAE																	
Gizzard shad	-	-	-	-	-	-	-	-	-	-	-	-	6	-	-	6	>1
ESOCIDAE																	
Grass pickerel	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1	>1
CYPRINIDAE																	
Stoneroller	-	-	-	1	2	1		25	54	1			29	16		129	6
Goldfish	8	-	-	-	-	1	1	-	-	-	-	5	-	-	-	15	1
Carp	2	8		40	-	-	4	2	-	1	-	8	2	-	-	67	3
Hornyhead chub	-	-	-	1	-	-	-	-	-	-	-	3	10	2	-	16	1
Golden shiner	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	1	>1
Emerald shiner	-	-	-	1	-	-	-	-	-	-	-	-	-	-	-	1	>1
Striped shiner	-	-	-	3	-	-	-	-	-	-	-	-	1	-	-	4	>1
Common shiner	-	-	-	-	7	1	-	-	-	-	-	-	-	-	-	8	>1
Bigmouth shiner	-	-	-	21	45	-	30	29	4	-	-	-	-	-	1	130	6
Red shiner	-	-	-	-	-	-	-	-	-	-	-	4	1	-	-	5	>1
Spotfin shiner	-	-	-	16	32	11	170	6	3	-	29	18	10	7	5	307	15
Sand shiner	-	2	-	7	30	14	2	-	-	-	2	-	8	9	-	74	4
Redfin shiner	-	-	-	-	34	-	2	18	-	-	-	2	-	-	-	56	3
Suckermouth minnow	-	-	-	-	-	-	-	-	-	-	-	-	16	2	-	18	1
Bluntnose minnow	-	5	3	89	45	47	78	57	12	-	1	1	2	9	61	410	20
Fathead minnow	4	-	43	4	-	-	1	1	1	-	-	-	-	-	-	54	3
Bullhead minnow	-	-	-	-	-	-	-	-	-	-	-	-	-	-	4	4	>1
Creek chub	16	2	-	6	1	-	-	10	42	-	-	-	-	-	-	77	4
Carp X Goldfish	-	-	-	-	-	-	-	-	-	-	-	-	2	-	-	2	>1
CATOSTOMIDAE																	
White sucker	4	1	-	2	-	-	-	3	9	-	-	-	-	-	-	19	1
Northern hogsucker	-	-	-	-	-	-	-	-	-	-	-	3	-	-	-	3	>1
Black redhorse	-	-	-	-	-	-	-	-	-	-	-	-	1	-	-	1	>1
ICTALURIDAE																	
Black bullhead	-	-	3	2	-	-	-	-	1	-	-	-	-	-	-	6	>1
Stonecat	-	-	-	-	-	-	1	-	-	-	1	1	2	-	-	5	>1
Tadpole madtom	-	-	-	-	2	-	-	-	-	-	-	-	-	-	-	2	>1
CYPRINODONTIDAE																	
Blackstripe topminnow	-	-	-	-	2	1	-	-	-	-	-	-	-	-	-	3	>1
CENTRARCHIDAE																	
Rock bass	-	-	-	-	-	-	1	-	-	-	-	1	-	-	1	3	>1
Green sunfish	17	7	18	20	1	2	54	102	11	7	-	60	8	24	76	407	20
Pumpkinseed	-	-	-	-	-	-	12	11	-	-	-	7	-	-	7	37	2
Orangespotted sunfish	-	-	-	-	-	-	-	-	-	-	-	3	-	-	6	9	>1
Bluegill	-	-	34	4	1	-	3	-	-	-	-	15	-	-	36	93	5
Longear sunfish	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	2	>1
Smallmouth bass	-	-	-	-	-	1	-	-	-	-	-	6	-	-	-	7	>1
Largemouth bass	-	-	-	-	-	-	-	-	-	-	-	4	4	-	-	8	>1
Black crappie	-	-	-	1	-	-	-	-	-	-	-	-	1	-	-	2	>1
Green sunfish X Pumpkinseed	-	-	-	1	-	-	2	-	-	1	-	8	-	-	3	15	1
Green sunfish X Longear sunfish	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	>1
Pumpkinseed X Bluegill	-	-	-	-	-	-	-	-	-	-	-	1	-	-	-	1	>1
PERCIDAE																	
Johnny darter	-	-	-	-	4	7	-	-	-	-	-	-	-	-	-	11	>1
TOTAL NUMBER OF INDIVIDUALS	51	25	101	223	206	86	361	264	137	10	33	151	103	69	203	2023	96
TOTAL NUMBER OF SPECIES	6	6	5	17	13	10	13	11	9	4	4	17	15	7	10	37	-
DIVERSITY (d)	1.90	2.01	1.65	2.81	2.74	2.01	2.19	2.37	2.17	0.99	1.50	3.13	3.14	2.13	2.29	-	-
EQUITABILITY (e)	0.83	0.92	0.80	0.56	0.69	0.55	0.13	0.64	0.67	0.63	1.00	0.66	0.78	0.79	0.65	-	-

APPENDIX D

LOCATION OF 1976 ILLINOIS ENVIRONMENTAL PROTECTION AGENCY WATER QUALITY SAMPLING SITES ON THE DUPAGE RIVER, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS

Basin Code	Station Number	Stream Name	Description
DuPage River			
GB	01	DuPage River	Old US 6 Br at SW edge Channahon
GB	02	DuPage River	New US 6 Br at NW edge Channahon
GB	03	DuPage River	Co Rd Br 1.5 mi NNW of Channahon
GB	04	DuPage River	TWP Rd Br 1 mi S of Shorewood
GB	08	DuPage River	Renwick Rd Br 1.5 mi SW Plainfld
GB	09	DuPage River	RT 59 Br at Plainfield
GB	10	DuPage River	Plainfield-Naperville Rd Br
Illinois and Michigan Canal			
GBE	02	IL & MI Canal	US 6 Br at Channahon
Lilly Cache Creek			
GBE	01	Lilly Cache Creek	US 30 Br 1 mi SE Plainfield
Norman Drain			
GBH	01	Norman Drain	Rt 59 Br at 143rd St Plainfield
DuPage River - West Branch			
GBK	01	W Br DuPage River	Hobson Rd Br at Naperville
GBK	02	W Br DuPage River	Washington St Rd Br S Naperville
GBK	03	W Br DuPage River	US 34-9th Ave Br at Naperville
GBK	04	W Br DuPage River	Warrenville Rd Br at Warrenville
GBK	05	W Br DuPage River	Rt 56-Butterfield Rd Br Warrenvl
GBK	06	W Br DuPage River	Mack Rd Br N of Warrenville
GBK	07	W Br DuPage River	Garys Mill Rd Br S of W. Chicago
GBK	09	W Br DuPage River	Rt 64 St Charles Rd Br N of W Chicago
GBK	10	W Br DuPage River	Br at Arlington 1 mi SW Hanover Pk
Spring Brook			

GBKA	01	Spring Brook	Winfield Rd Br 1 mi N of Warrenvl

Kress Creek

GBKB	01	Kress Creek	Rt 59 Br S of West Chicago
GBKB	03	Kress Creek	Townline Rd Br SW of W Chicago
GBKB	04	Kress Creek	McChesney Rd Br S of Roosevelt Rd
GBKB	05	Kress Creek	Hawthorne W Br at NW edge W Chicago

DuPage River - East Branch

GBL	01	E Br DuPage River	Hobson Rd Br 2 mi S of Lisle
GBL	02	E Br DuPage River	Washington St Rd Br 1 mi S Naperville
GBL	05	E Br DuPage River	Maple Ave Br at Lisle
GBL	07	E Br DuPage River	Rt 56-Butterfield Rd Br
GBL	08	E Br DuPage River	Rt 38-Rossevelt Rd Br at Glen Ellyn
GBL	09	E Br DuPage River	Rt 64-North Ave Br Nr Glendale HTS

APPENDIX E

MAXIMUM, MINIMUM, MEAN, AND NUMBER OF ANALYSES (IN PARENTHESES) FOR PARAMETERS (BASED UPON ILLINOIS ENVIRONMENTAL PROTECTION AGENCY WATER QUALITY NETWORK FOR 1976) CONTRIBUTING SIGNIFICANTLY TO THE TOXICITY INDEX FOR SITES ON THE DUPAGE RIVER, COOK, DUPAGE, AND WILL COUNTIES, ILLINOIS

IEPA Station	Amm-N (mg/l)	CN (mg/l)	MBAS (mg/l)	Ag (mg/l)
DuPage River - West Branch				
GBK10	.12 - 13.00 3.95 (9)	.000 - .010 .003 (6)	.60 - .60 .60 (2)	.000 - .000 .000 (6)
GBK09	.09 - 1.40 .52 (8)	.000 - .000 .000 (3)	.20 - .60 .40 (2)	.000 - .000 .000 (3)
GBK07	.42 - 5.10 2.13 (8)	-- --	-- --	-- --
Kress Creek				
GBKB05	2.60 - 26.00 9.43 (8)	.000 - .000 .000 (3)	.20 - .20 .20 (2)	.000 - .000 .000 (3)
GBKB04	.76 - 3.30 1.77 (7)	-- --	-- --	-- --
GBKB03	.01 - 3.20 .91 (6)	.000 - .000 .000 (3)	.30 - .30 .30 (1)	.000 - .000 .000 (3)
GBKB01	.06 - 2.50 .69 (7)	.000 - .000 .000 (3)	.20 - .50 .35 (2)	.000 - .000 .000 (3)
DuPage River - West Branch				
GBK06	.30 - 3.50 1.12 (7)	.000 - .000 .000 (3)	.20 - .20 .20 (1)	.000 - .000 .000 (3)
Spring Brook				
GBKA01	4.20 - 18.00 9.25 (8)	-- --	.70 - .70 .70 (1)	-- --
DuPage River - West Branch				
GBK05	.72 - 6.40 3.05 (7)	.000 - .010 .003 (3)	.40 - .80 .60 (2)	.000 - .000 .000 (3)

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IEPA Station	Amm-N (mg/l)	CN (mg/l)	MBAS (mg/l)	Ag (mg/l)
DuPage River - West Branch (continued)				
GBK04	.65 - 9.00	--	--	--
	3.05 (8)	--	--	--
GBK03	.35 - 8.00	.000 - .000	.20 - .20	.000 - .000
	1.89 (8)	.000 (5)	.20 (1)	.000 (5)
GBK01	.06 - 7.20	--	--	--
	1.32 (8)	--	--	--
GBK02	.02 - 6.40	.000 - .010	.60 - .70	.000 - .000
	1.40 (8)	.003 (3)	.65 (2)	.000 (3)
DuPage River - East Branch				
GBL09	2.20 - 13.00	.010 - .010	1.40 - 1.40	.000 - .000
	6.23 (6)	.010 (1)	1.40 (1)	.000 (1)
GBL08	4.10 - 17.00	--	--	--
	8.43 (6)	--	--	--
GBL07	2.80 - 16.00	.000 - .020	.70 - .90	.000 - .000
	7.20 (7)	.010 (5)	.80 (2)	.000 (5)
GBL05	1.50 - 12.00	.010 - .020	.60 - 1.00	.000 - .020
	4.97 (8)	.017 (3)	.80 (2)	.007 (3)
GBL01	1.40 - 13.00	.000 - .010	.40 - .40	.000 - .000
	5.51 (8)	.002 (5)	.40 (1)	.000 (5)
GBL02	.76 - 10.00	--	--	--
	3.43 (8)	--	--	--
DuPage River				
GB10	7.00 - .45	.000 - .000	.20 - 20	.000 - .000
	2.57 (7)	.000 (4)	.20 (1)	.000 (4)
Norman Drain				
GBH01	.00 - .27	--	--	--
	.10 (4)	--	--	--
DuPage River				
GB09	.02 - .66	.000 - .000	.30 - .60	.000 - .000
	.28 (4)	.000 (2)	.45 (2)	.000 (2)
GB08	4.60 - .01	.000 - .000	.60 - .60	.000 - .000
	1.10 (5)	.000 (2)	.60 (2)	.000 (2)
Lilly Cache Creek				
GBE01	.00 - .75	.000 - .000	.20 - .40	.000 - .000
	.22 (5)	.000 (2)	.30 (2)	.000 (2)

EPA-287 (Cin.)
(4-76)

IEPA Station	Amm-N (mg/l)	CN (mg/l)	MBAS (mg/l)	Ag (mg/l)
DuPage River				
GB04	.00 - 1.20 .37 (5)	.000 - .000 .000 (1)	— —	.000 - .000 .000 (1)
GB03	.00 - .60 .14 (5)	.000 - .000 .000 (2)	.40 - .60 .50 (2)	.000 - .000 .000 (2)
Illinois and Michigan Canal				
GBA02	.00 - .95 .23 (5)	.000 - .000 .000 (2)	.40 - .40 .40 (1)	.000 - .000 .000 (2)
DuPage River				
GB02	.00 - 1.10 .35 (5)	.000 - .000 .000 (2)	.60 - .60 .60 (1)	.000 - .000 .000 (2)
GB01	.00 - 1.00 .33 (5)	.000 - .000 .000 (1)	— —	.000 - .000 .000 (1)
IEPA Standards	1.50	.025	—	.005

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